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Effect of converting row crop to prairie on nutrient concentration in shallow groundwater and soil properties

Bethany Brittenham
Iowa State University

Effect of converting row crop to prairie on nutrient concentration in shallow groundwater and soil properties

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Bethany Anne Brittenham

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Program of Study Committee:
Matthew J. Helmers, Major Professor
Daniel S. Andersen
Randall K. Kolka
Mark D. Tomer

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ABSTRACT

Alteration of the Iowa landscape transformed millions of hectares of tallgrass prairie into highly productive fields of primarily corn (Zea mays L.) and soybean [Glycine *max*. (L.) Merr.]. Introduction of native prairie in contour strips and at the footslope within row crop fields has been shown to reduce nutrient export from fields thus reducing adverse environmental effects. Inclusion of prairie within row crop fields provided an opportunity to modify soil properties to similar conditions prior to row crop use via organic matter addition and deep rooting.

Nutrient concentrations in shallow groundwater beneath row crop have been shown to be elevated compared to nutrient concentrations in shallow groundwater beneath native vegetation. The first study detailed in this thesis compared concentrations of nitrate-nitrogen and phosphorous in groundwater beneath four treatments: 100% row crop, 10% footslope prairie strip (PS), 10% contour coupled with footslope PS, and 20% contour coupled with footslope PS. Maximum annual nitrate-nitrogen fluxes (kg ha^{-1}) in the top 2 m beneath the soil surface in order from largest to smallest were 100% row crop, 10% foot slope PS, 10% PS in contours with footslope cover, and 20% PS in contours with footslope cover. Maximum annual fluxes (kg ha^{-1}) for phosphorous were in decreasing order 10% footslope PS, 10% contour with footslope cover, 100% row crop, and 20% contour with footslope cover. In the 100% row crop treatment, it was possible phosphorous was exported with runoff instead of deposited with sediment at footslopes with phosphorous-releasing conditions.

The second study reviewed soil data collected from 6 sites in 5 distinct locations throughout Iowa. A subset from sites with similar soil types was reviewed to determine the effects of reversion to native prairie from row crop for a chronosequence of 0, 10, 25, and 37 years. The remaining 3 sites with differing soil types were analyzed for 0 and 2 year trends. Soil properties measured from all sites were total nitrogen (TN), total carbon (TC), pH, bulk density, aggregate size distribution, and particulate organic matter (POM) associated carbon and nitrogen. In general, both carbon and nitrogen increased while maintaining a similar TC:TN. Bulk density decreased with time and pH did not follow a distinct pattern. After 10 years in prairie, macroaggregate fractions increased significantly and were maintained over time. Carbon and nitrogen content within aggregate fractions increased significantly while maintaining the TC:TN ratio. Within the POM fractions, TC and TN did not express a general increasing trend though the TC:TN ratio increased. Conservatively, prairie litter and dead roots annually provided 1950 kg C ha^{-1} and 2250 kg C ha^{-1} more than corn/soybean and continuous corn rotations, respectively. Annually prairie litter contained 53 kg N ha^{-1} and 57 kg N ha^{-1} more than corn/soybean rotation and continuous corn, respectively.

High variability in soil texture, soil genesis, and precipitation patterns warrant further investigation into both shallow groundwater and soil property alteration following conversion from row crop to prairie. Further study will assess the applicability of integrating prairie vegetation as a wide-spread conservation practice.

CHAPTER 1. GENERAL INTRODUCTION

Background

This thesis contains water and soil data collected as a part of the Science-based Trials of Rowcrops Integrated with Prairie Strips (STRIPS) project at Iowa State University. A multidisciplinary project, implementation of STRIPS began at the Neal Smith National Wildlife Refuge in 2007 where prairie strips (PS) were integrated into row crop fields of corn (Zea mays L.) and soybean [Glycine *max*. (L.) Merr.]. With numerous collaborators, institutional partners, funders, and stakeholder advisory committee members, this project aims to combine knowledge and resources in pursuit of sustainable practices for reducing ecological and hydrological effects of row crop fields on the environment. Three key questions were posed at the inception of the STRIPS project:

1. What is the capacity for multifunctional benefits (e.g. water purification, recreational opportunities, and bio-diversity conservation)?
2. Does placement of prairie vegetation affect capacity for multifunctional benefits (e.g. slope, contour vs edge-of-field PS placement)?
3. What is the threshold percent conversion from row crop to prairie necessary for multifunctional benefits (Schulte, et al., 2006)?

In recent years, the STRIPS project expanded to include 6 paired research watersheds and over 30 private landowners with implemented sites throughout the state of Iowa. Continued interest in conservation practices across the agricultural landscape necessitates the availability of science-based materials (Lovell and Sullivan, 2006) and supports the need for long-term research studies. Implementation of grassed waterways, erosion prevention strategies, and nutrient management aim to reduce nonpoint source pollution from production agriculture, but contamination of shallow groundwater by nutrients persists. Nitrogen leaching into groundwater is well documented and dissolved phosphorous resulting from sediment deposition is becoming more understood (Stutter, et al., 2009; Tomer, et al., 2010). The opportunity to retain nutrients within the row crop field, or at least reduce their export could be imperative to reducing the impact of chronic conditions like hypoxia in the Gulf of Mexico (Foley, et al., 2005; Robertson and Vitousek, 2009).

With the addition of PS in the landscape, it is reasonable to expect a change in soil properties beneath the perennial vegetation. Previous studies suggest the conversion from row crop to prairie results in carbon and nitrogen accumulation (Anderson-Teixeira, et al., 2009; Breuer, et al., 2006; Knops and Tilman, 2000), an increase in infiltration accompanied by larger soil aggregates and reduced erosion (Bharati, et al., 2002; Le Bissonnais, 1996). Accumulation of nutrient-rich macroaggregates and particulate organic matter may increase the potential for nutrient cycling over time (Elliott, 1986).

A chronosequence of soil structure following conversion from row crop to prairie aids in determining likely changes associated with the introduction of perennial land cover. As soil properties change following conversion to prairie land cover, we would expect similar changes within PS. However, soil properties like aggregate size and nutrient content are related to soil texture and climate. Thus, chronosequence comparisons should be made locally or regionally when data is available. The objectives of this thesis are to:

1. Investigate how percent conversion from row crop to PS affects nitrate-N and phosphorous concentration as well as flux in shallow groundwater.

2. Quantify changes in soil properties for a prairie restoration chronosequence in Central Iowa.

Thesis Organization

Chapter 2 fulfills objective 1 providing insight on nitrate-N and phosphorous concentration in groundwater under 100% row crop fields as well as fields treated with varying percentages and layouts of PS at the Neal Smith National Wildlife Refuge near Prairie City, Iowa. This includes nutrient flux estimations from the catchments as well as a water balance to quantify the magnitude of water subjected to chemical and physical interactions with the prairie strip treatments. Chapter 3 summarizes soil parameters from 6 sites within Iowa at 0, 2, 10, 25, and 37 years post conversion from row crop to prairie. The resulting chronosequence of soil health measurements provides an estimation of expected changes in soil parameters following the land use change. Chapter 4 summarizes general conclusions from this thesis and suggestions for future research on the integration of prairie vegetation to a row crop landscape. References, figures, and tables follow their corresponding chapter.

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CHAPTER 2. NITRATE AND PHOSPHOROUS DYNAMICS IN SHALLOW GROUNDWATER WITH PRAIRIE STRIPS

A paper to be modified for submission to *Journal of Environmental Quality*

Bethany A. Brittenham[1], Christopher C. Witte, Randy K. Kolka, Mark Tomer, Matthew J. Helmers

Abstract

Prairie strips (PS) integrated within a row crop field with no-till, no-tile corn-soybean rotation have been shown to reduce nitrate-nitrogen (NO_3-N) contamination in shallow groundwater over a short-term period in contrast to a fully cropped field. Additionally, phosphorous (P) was often overlooked as a groundwater contaminant due to its low leaching capacity and strong sorption to soils. However, the important role of P in production agriculture led to consideration as a potential groundwater contaminant. Thus, the objective of this study was to determine how effective PS were at preventing contamination in shallow groundwater from both NO_3-N and dissolved P in a row crop field for the study period of 2007-2016. In this study, there were twelve catchments in four blocks with four randomly assigned treatments: 100% row crop, 10% PS in contour strips and at the footslope, 10% PS at the footslope, and 20% PS in contour strips and at the footslope. Prairie strips differed from typical vegetative buffers since they consisted of native prairie species and are incorporated among cropped rows as well as at the field edge. Nitrate-N concentration in shallow groundwater at the footslope for the 2007-2016 time

[1] Primary author

interval was significant by treatment from highest to lowest as: 100% row crop, 10% PS contour strips, 10% PS footslope cover, and 20% PS contour strips. Phosphorous concentrations were highest at 10% PS footslope cover sites. Both 20% and 10% PS in contours had similar P concentrations. However, 10% PS in contour strips did have significantly higher P concentrations compared to the 100% row crop cover ($p<0.05$). An estimation of groundwater flux for the May through October growing season indicated the 100% row crop treatment exported significantly more NO_3-N than the PS treatments, and the 10% PS at the footslope exported significantly more P ($p<0.05$). Results from this study may aid in the selection of PS as a conservation practice for nutrient reduction in shallow groundwater as well as inform management decisions for PS layout on the landscape.

Introduction

Interest in the effect of agricultural production on hydrologic systems coupled with increasing pressure to address environmental concerns such as eutrophication emphasize the need to develop a detailed review of conservation practices available to producers (Schmitt, et al., 1992). Conversion of native prairie to farmed land reduces natural nutrient management processes and increases agriculture-associated pollutants such as sediment and nutrients in surface and shallow groundwater (Hernandez-Santana, et al., 2013; Strebel, et al., 1989; Turner and Rabalais, 2003; Zhou, et al., 2010). Interest from producers requires the availability of science-based materials to inform decisions (Lovell and Sullivan, 2006). Current field-scale work on integration of perennial filter strips within row crop acres better informs the decision-making processes for one of these practices (Dorioz, et al., 2006; Hernandez-Santana, et al., 2013).

Decline in perennial prairie vegetative cover in favor of highly productive row crop systems in the Upper Midwest contributes directly to surface water quality impairments and chronic conditions such as hypoxia in the Gulf of Mexico (Foley, et al., 2005; Robertson and Vitousek, 2009). In the state of Iowa, less than 1% remains of the historical 12.5 million hectares of tallgrass prairie (Samson and Knopf, 1994; Smith, 1990). Current emphasis on corn (Zea mays L.) and soybean [Glycine *max*. (L.) Merr.] production results in minimal winter and early spring vegetative cover and a high priority on maximum production efficiency during the growing season. As a result, natural buffering processes are eliminated as well as capacity for infiltration, retention, and percolation of precipitation resulting in more runoff (Hernandez-Santana, et al., 2013; Zhang and Schilling, 2006).

Implementation of grassed waterways and other conservation practices aids in some field erosion prevention and nutrient loss reduction, but nonpoint source pollution in surface water and shallow groundwater persists. In response to increasing environmental concerns, landscape restoration including the installation of contour buffer strips within the field and riparian buffer strips at field edges gained interest as practices to lessen nutrient and sediment transport out of the system. These practices intersect flow paths and provide a final barrier to encourage nutrient processing. Unlike prairie strips (PS), these practices do not highlight the utilization of native prairie species (NRCS, 2007; 2011). For instance, the NRCS standard for contour buffer strips emphasizes protective covering during "critical erosion periods" (NRCS, 2011). This may emphasize cool-season grasses that are viable during moist springs for nutrient and water uptake prior to row crop establishment.

PS may be comprised of a variety of vegetation including native prairie forbs and grasses (Lovell and Sullivan, 2006). In contrast to traditional edge-of-field buffer and contour strips, PS may be placed throughout the landscape including footslopes and along hillsides to lessen flow carrying capacity and increase natural nutrient processing (Zhang and Schilling, 2006). Analysis of varying percentages of row crop fields converted to PS, coupled with strategic placement methods, provides a review of one strategy for nutrient removal (Hernandez-Santana, et al., 2013; Pérez-Suárez, et al., 2014; Schmitt, et al., 1992; Zhou, et al., 2010). The research described herein builds off a short-term study conducted at the same site emphasizing the need for continued monitoring of shallow groundwater following implementation of perennial filter strips in central Iowa to better inform stakeholders on the long-term impact of PS (Zhou, et al., 2010).

Monitored shallow groundwater provides 10 years of data post-conversion to PS for extended reference. Nitrate-N data through 2008 was previously published (Zhou, et al., 2010) and included here for completeness. As with any conservation system, one year post PS implementation may show little to no significant change in contaminant levels (Zhou, et al., 2010), and a long-term study is more likely to display the effect of treatments (Novak, et al., 2000). The objectives of this study were to quantify the effects of PS implementation on (i) NO_3-N and P in shallow groundwater from no-till, no-tile agricultural fields at the Neal Smith National Wildlife Refuge (NSNWR) in Central Iowa.

Materials and Methods

Site Description

This study was conducted at the Neal Smith National Wildlife Refuge (NSNWR; 41˚33'N; 93˚16'W) in Jasper County, Iowa (Figure 2.1). Managed by the U. S. National Fish and Wildlife Service since its inception in 1990, the refuge is converting the landscape back to native prairie. While awaiting reconstruction, areas of the refuge are kept in bromegrass (*Bromus* L.) or leased for row crop production under management guidelines set by the Fish and Wildlife Service.

A part of the Pre-Illinoian Southern Iowa Drift Plain (Prior, 1991), the region included rolling hills with abundant groundwater (NRCS, 2006). Dominated by mollisols, uplands are primarily arguidolls with steeply sloping hapludalfs (NRCS, 2006). The NSNWR's landscape has been incised by ephemeral and perennial streams. Precipitation for the site was summarized from the MesoWest station approximately 1 km north and west of the Interim site on the refuge (Figure 2.1). Annual precipitation for the reporting period (2007-2016) averaged 970 mm (Figure 2.2).

Experimental Design

Twelve zero-order (ephemeral flow) catchments were used for the balanced incomplete block design with four blocks and three treatments per block: Basswood (two blocks), Interim (one block), and Orbweaver (one block). Treatments were 100% row crop, 20% PS in contour and footslope strips, 10% PS in contour and footslope strips, and 10% at the footslope only (Figure 2.3). Prior to modifications for this study, all sites were in bromegrass (*Bromus* L.) for a minimum of 10 years. Catchments varied in size from 0.47 to 3.19 ha with an average slope range of 6.1 to 10.5% (Gutierrez-Lopez, et al., 2014; Hernandez-Santana, et al., 2013; Zhou, et al., 2010). Table 2.1 details the relative texture and treatments of the twelve catchments (adapted from J. Gutierrez-Lopez et al., 2013).

In August 2006, sites were tilled to accommodate spring 2007 planting to a corn-soybean rotation with corn planted even years. Prairie strip were broadcast seeded in July of 2007 with a mix of over 20 species primarily composed of aster (*Aster* L.), big bluestem

(*Andropogon gerardii* Vitman), little bluestem (*Schizachyrium* Nees), and indiangrass (*Sorghastrum* Nash) (Hirsh, et al., 2013). Prairie strip width varied from 27 to 41 m at footslope and 5 to 10 m in the contour strips. Prairie strip percentage accounted for treatment designation, not strip area as catchment size directly affected land area converted to PS. Sites were not artificially drained, and no regular tillage was scheduled with maintenance practices. Anhydrous ammonia was injected into the field at rates indicated in Table 2.2. Supplemental MAP fertilizer was applied as needed in the spring prior to corn planting, and tillage to remove gullies was performed sparingly (Table 2.2).

Shallow Groundwater Wells

At the footslope positions, each catchment had one shallow groundwater well (Figure 2.3) sealed with bentonite grout at the ground level to prevent runoff from directly entering the groundwater. Each well was 50 mm PVC with 0.6 m screens and at a depth between 2.9 and 5.4 m. Samples were collected with a hand pump and tubing lowered to the bottom of the well. Prior to collection, wells were purged until empty and allowed to recharge (0.5-24 hours). Sampling began in 2006 and occurred monthly from April to October for most years.

Nutrient Analysis

Groundwater samples were analyzed for combined nitrate-nitrogen and nitrite-nitrogen (hereto referred to as NO_3-N) from 2006-2016 with the AQ2 method EPA-114-A. Samples above 15 mg L^{-1} were diluted until they were within the 0.25 to 15 mg L^{-1} range. Concentrations below 0.25 mg L^{-1} were then analyzed with the AQ2 method EPA-127-A. For phosphorous (P), the AQ2 method EPA-118-A was utilized.

The 2006-2010 and 2014-2016 groundwater samples were filtered with a 0.45 μm filter (DS0210 membrane filter, Nalgene Labware, Rochester, NY) resulting in a measurement of dissolved reactive phosphorous. The 2006-2010 samples were analyzed with the Lachat QuikChem 8000 Flow Injection Analyzer (Loveland, CO). The minimum standard for NO_3-N was 0.25 mg L^{-1} with a minimum detection of 0.01 mg L^{-1}, and the minimum standard for P was 0.005 mg L^{-1} with a detection limit of 0.001 mg L^{-1}. All samples were stored at 4˚C prior to analysis.

Beginning in 2011, instrumentation for analyses and thus standards and detection limits changed. The minimum standard for NO_3-N became 0.012 mg L^{-1} with a limit of detection at 0.003 mg L^{-1}. For P, the lowest standard became 0.01 mg L^{-1} with a detection limit of 0.002 mg L^{-1}. Groundwater samples 2011-2013 and 2015 were not filtered for a measurement of total reactive phosphorus. The 2011-2016 analysis utilized a Seal Analytical AQ2 Discrete Autoanalyzer (Mequon, WI). Samples were stored at 4˚C while awaiting analyses.

As noted, 2015 samples were analyzed both prior to and post filtering to determine if there was a quantifiable reduction in P following filtering. Any reductions were negligible. A regression equation describing the relationship between dissolved (filtered) and total reactive (unfiltered) P for 2015 indicated an almost 1:1 ratio between the two forms (Figure 2.4). The coefficient of determination equals 0.995 with a standard error of 0.0063. Thus, annual P concentrations were deemed comparable regardless of whether the groundwater samples were filtered.

Statistical Analyses

The nutrient datasets contained censored values due to analysis limitations. Thus, the LIFEREG procedure in SAS software was utilized to model in censored values both below detection and below the minimum standard (SAS, 2012). Analysis of variance was conducted through LIFEREG to determine the statistical difference between each treatment both on annual and total interval basis (2007-2016). For the depth to groundwater measurements and flux calculations, the GLIMMIX procedure was used for analysis of variance. For all analyses, blocking was incorporated to separate Basswood 1 through 3 and Basswood 4 through 6 into separate blocks to round out a balanced incomplete block design. Repetition in PS treatments were treated as replicates. To determine if the NO_3-N concentration was leveling off in the 100% row crop treatments, the GLM procedure was used to check for a strong temporal trend for the 2013-2016 NO_3-N data.

Estimating Nutrient Flux

By modifying Darcy's Law to account for an unconfined aquifer with sloping bottom, Equation 1 described the area-weighted flux at the footslope for nutrients leaving the catchments:

$$J_{nutrient} = \left(\frac{\left(-K * w * \left(\frac{h_2 - h_1}{\Delta x} * \frac{h'_1 + h'_2}{2} \right) h' * C \right)}{A} \right) \tag{1}$$

where K was the saturated conductivity (m d^{-1}) estimated by particle size (Tietje and Hennings, 1996), w was the average width of the watershed (m) based on area and estimated length (Zhou, et al., 2010), x was the distance between the summit and footslope wells (m), h_1 was the height of the summit water table at mean sea level (m), h_2 was the height of the footslope water table at mean sea level (m), h'_1 was the difference between height of the summit water table and effective depth (m), h'_2 was the difference between height of the footslope water table and effective depth (m), C was the nutrient concentration (kg m^{-3}), and A was the catchment area (m^2). Shallow groundwater depth was defined as 2 meters beneath the soil surface and assumed to be the maximum depth of substantial root interaction and denitrification (Weaver, 1958; Weaver, et al., 1935).

Nutrient flux within each catchment was calculated utilizing the measured groundwater depths for the May through October growing season each year. This flux output quantified the amount of nutrient exported from the catchment via the groundwater 2 meters below the soil surface based on direct measurements.

Water Balance Estimation

A simple water balance (Equation 2) provided an estimation for water infiltrating past the 2 m shallow groundwater zone

$$I_u = P - RO - I_t - ET \quad (2)$$

where I_u was the untreated infiltration or deep flow (cm). P was precipitation (cm) measured by the NOAA station near the Interim site, and RO was runoff applied as a depth measurement (cm) over the watershed. I_t was shallow infiltration within the 2 m depth (cm) calculated from Equation 1 reported as depth over the catchment area. ET was evapotranspiration (cm) estimated by similar studies and applied by crop type and percentage in each catchment (Bakhsh, et al., 2004; Brye, et al., 2000; Mateos Remigio, et al., inpreparation). Evapotranspiration (ET) for prairie, soybean, and corn, were 44, 40, and 41 cm respectively for the 6 month growing season (May-October). In watersheds with varying land cover, ET was weighted by area. Thus, this water balance applied to a 6 month growing season.

Results

Groundwater Fluctuation

Depth to shallow groundwater for the 6 month growing season varied from approximately 0.1 to 3.5 meters below the ground surface for the 2006-2014 time period (Figure 2.5). Groundwater depth measurements were not taken for 2015 and 2016. Groundwater levels tended to be closest to the soil surface in the spring and increased in depth through the summer until fall when depths began decreasing.

The largest total variance across all treatments in groundwater depth for the May-October growing season occurred in 2012 (Table 2.3). For the 2007-2014 growing season, variance in depth to groundwater for 10% PS at the footslope and 100% row crop treatments was significantly greater than the 20% PS in contours (Table 2.3). There was no significant difference in groundwater depth variance between the 10% PS in contour treatment compared to the 100% row crop, 10% PS at the footslope and 20% PS in contour strip treatments.

Nitrate-Nitrogen Concentration in Shallow Groundwater

Following conversion to row crop, NO_3-N concentrations at the summit wells within catchments increased most noticeably under the 100% row crop and 10% PS footslope cover (Figure 2.6). Yearly comparison of nutrient concentration for the 2007-2016 period indicated treatments without PS in contour strips (100% row crop and 10% PS at the footslope) had significantly higher concentrations of NO_3-N compared to the 10%

and 20% PS in contour strips (Table 2.4). Additionally, 10% PS in contour strips expressed significantly higher concentrations of NO_3-N than 20% PS in contour strip treatments.

Footslope NO_3-N concentration increased most prominently in the 100% row crop treatment 10 years post conversion from row crop to prairie (Figure 2.7). Typically, the 10% footslope cover and 20% PS in contours presented similar NO_3-N concentrations in contrast to the other two treatments. Analysis revealed a significant difference between all treatments ($p<0.05$) for the total 2007-2016 time interval (Table 2.5). The highest to lowest NO_3-N concentration in shallow groundwater by treatment were 100% row crop, 10% PS contour strips, 10% PS footslope cover, and 20% PS contour strips. The presence of any of the PS treatments reduced NO_3-N concentrations by 77% compared to the 100% row crop treatment. During the 2013-2016 time frame, Figure 2.7 appeared to depict the NO_3-N concentration leveling off for the 100% row crop treatments. Considering a statistical effect for site, the trend for the years of interest (2013-2016) is not significant ($p=0.300$).

Phosphorous Concentration in Shallow Groundwater

Following conversion to row crop, P concentrations by treatment at the summit wells within catchments did not follow a distinct trend (Figure 2.8). Comparison between years by treatment indicated the summit groundwater beneath the 10% PS footslope treatment contained significantly higher levels of P than the 100% row crop, 10% PS in contour strips, and 20% PS in contour strips.

Phosphorous concentrations during the 2007-2016 time interval increased most at the 10% footslope cover sites (Figure 2.9) for a significantly different ($p<0.05$) value compared to the remaining treatments (Table 2.7). There was no significant difference between 20% PS in contours compared to 10% PS in contours and 100% row crop.

However, 10% PS contour strips and 100% row crop were significantly different ($p<0.05$). There did not appear to be a temporal trend for P in groundwater at the footslope of treatments (Figure 2.9). The large spikes in P concentration of 2011 and 2012 followed years with high runoff events (unpublished data) which may indicate substantial sediment deposition at the footslope.

Nutrient Flux

Nitrate-nitrogen flux on a per hectare basis for the May-October growing season indicated the inclusion of PS into the landscape significantly ($p<0.05$) reduced annual export of NO_3-N compared to 100% row crop treatments (Table 2.8). Significant differences were apparent as early as 1 year post-conversion.

Phosphorous flux did not appear to exhibit a temporally increasing trend. For the 6 month growing season and the 10 years of data, 10% PS at the footslope exported significantly more P than all PS treatments (Table 2.9). There was no significant difference in P export for the 100% rowcrop, 10% PS in contours, and 20% PS in contours treatments.

Water Balance

The water balance (Figure 2.10) showed the largest usage of water occurred from plant uptake in all years, excluding the extreme precipitation of 2010 (Figure 2.2). The water balance also highlighted the disparity among treatments in terms of runoff. Basswood 4 site (10% PS in contours) frequently developed groundwater seeps during the experiment period. This contributed to the runoff quantity. In order from smallest to largest quantity regardless of treatment, the fluxes are as follows: shallow flux, runoff, deep flux, and ET.

Discussion

Groundwater Table

It has been shown that soil with perennial plant cover and higher evapotranspiration (ET) enabled greater infiltration rates than bare soil due to reductions in bulk density, macropore development, and plant water use (Bharati, et al., 2002; Zhang and Schilling, 2006). During the growing season, the PS likely insulated the soil's surface to lessen evaporation (Zhang and Schilling, 2006). However, much of that retained rainfall was then removed by perennial plant ET through the spring, summer, and fall (Schilling and Drobney, 2014). In contrast, row crops uptake soil water primarily in the summer months (Zhang and Schilling, 2006). Thus, we may expect less yearly variability in groundwater depth under perennial vegetation.

An additional factor for groundwater variance was highlighted by the greater annual runoff in the 20% PS treatment. The Basswood 4 catchment (20% PS treatment) frequently developed seeps that can be so extreme planting was delayed or impossible. As a result, the high runoff may actually be water from the unusual elevated groundwater. In general, footslope wells in catchments designated as 20% PS are at lower elevations than other treatment groups (Table 2.1). Thus, the seemingly elevated groundwater levels (low variability) may result from landscape position more than PS treatment. Additionally, precipitation will not infiltrate saturated ground and rainfall on that portion of the catchment would run off.

Nitrate-Nitrogen Concentration

As expected, summit NO_3-N concentrations in treatments without a contour PS component (100% row crop and 10% PS at the footslope) were statistically similar likely since no treatment was applied in the upslope positions. The 20% PS in contour strip treatment contained lower concentrations of NO_3-N than the 10% PS in contours, 10% PS at the footslope, and 100% row crop treatments likely given the summit implementation of PS which results in no fertilizer application (and associated leaching) at the PS location.

Mechanisms of NO_3-N reduction in groundwater were previously quantified for 100% row crop and 10% PS at the footslope catchments at NSNWR and indicated the primary method of NO_3-N removal was denitrification for an 137-day study (Mitchell, et al., 2015). Additional minor NO_3-N removal occurred as perennial vegetation uptake and incorporation into soil organic matter (Mitchell, et al., 2015; Perez-Suarez, et al., 2014). Overall, the presence of PS at the study catchments reduced NO_3-N concentrations disproportionately more than the percent of row crop converted to PS. This supported the disproportionate benefits hypothesis for the integration of perennials into agricultural landscapes (Asbjornsen, et al., 2013) in terms of hydrologic regulation and water quality.

Since the soil had high clay content, the lag prior to the 2008 spike in concentration is not unusual though it occurred prior to anhydrous ammonia application (Zhou, et al., 2010). The 2006 tillage may be responsible for the spike seen in 2008 following microbial mineralization of soil organic matter (Dinnes, et al., 2002). With a range in catchment lengths from 107 m, to 308 m, it is reasonable to expect quantifiable treatment effects in shallow groundwater at some catchments within 2 years.

Denitrification was enhanced by the shallow water tables that may be within 0.5 m from the surface. It has been shown that warm, wet springs increase soil nitrification, which coupled with the lack of crop present for NO_3-N uptake, promotes leaching into the groundwater (Dinnes, et al., 2002). Additional studies showed enrichment of soil organic carbon and dissolved organic carbon by PS served as a food source for denitrifying bacteria and the primary sink for NO_3-N leaving systems with PS treatments (Anderson-Teixeira, et al., 2009; Mitchell, et al., 2015).

Rainfall patterns also influenced shallow groundwater NO_3-N concentrations. The large spike in July of 2010 may be attributed to one-third of the annual average rainfall occurring the June before that sample. A similarly high rainfall in August of 2010 and decrease in NO_3-N concentration may have resulted from dilution following the June flush of NO_3-N into the groundwater (Dinnes, et al., 2002). For 2012 and 2013, total annual rainfall was below the expected annual average of 850 mm at 590 and 740 mm, respectively. This may have caused NO_3-N to accumulate within the soil profile (Dinnes, et al., 2002) and account for the apparent spike early in 2013. Continued periods of moderate to slightly high precipitation for the 2014-2016 interval at 870, 1010, and 900 mm, respectively may account for the seemingly steady-state concentrations of groundwater NO_3-N.

Presence of prairie vegetation in the catchment regardless of layout averaged 77% lower NO_3-N concentrations in the groundwater compared to the 100% row crop treatment. However, variation in rainfall quantity and seasonality across the state of Iowa indicates the need for further PS implementation and monitoring to assess the range in expected NO_3-N reductions in other regions. Mitchell et al. (2015) indicated the need for replication in different climate and hydrological settings to understand the impact of broader implementation. Overall, it appears that NO_3-N concentrations may be leveling off as has been seen in other agriculturally recharged groundwater systems (Strebel, et al., 1989). Continued monitoring will clarify if the trend is merely a result of precipitation and timing, or a new equilibrium for NO_3-N concentration in groundwater.

Phosphorous Concentrations

Summit P concentrations in shallow groundwater did not adhere to the expectation of statistically similar results for the 100% row crop and 10% PS at footslope treatments given the lack of summit PS application. The significantly greater concentration of summit groundwater P in the 10% PS footslope treatment may indicate higher soil P content in the 10% PS at footslope catchments. Thus, elevated soil P levels within catchments contributed to an elevated P concentration at the footslope wells.

Phosphorous at footslope wells with PS likely resulted from sediment accumulation within the perennial vegetation after transport via runoff (Tomer, et al., 2010). This small increase in available P may have been enough to induce leaching (Stutter, et al., 2009). Higher root density due to PS growth has also been shown to yield increased infiltration (Bharati, et al., 2002) and thus P transport to shallow groundwater due to macropore formation (Stutter, et al., 2009). Moreover, natural P removal processes do not occur as with NO_3-N. Thus, P removal from the system occurred primarily with biomass removal when the strips were harvested and removed from the site (Stutter, et al., 2009). Controlled burning in 2015 likely deposited P stored in plant tissue back onto the soil surface where saturated P conditions may already exist. A continuation of this study may help determine the effect of mowing versus burning PS on nutrient concentrations in shallow groundwater.

High precipitation events increase the potential for soil runoff and trapping within the PS resulting in an increase in potential for dissolved P transport to groundwater. At the footslope, shallow groundwater tables and anaerobic, denitrifying conditions produced an environment favorable for increasing P solubility by releasing iron-bound P (Tomer, et al., 2010) and releasing calcium-bound P (Browne, et al., 2008). The iron fixation of P may also be inhibited by the presence of sulfate produced by oxidation of iron sulfide by NO_3-N (Smolders, et al., 2009).

From the initial year of PS and row crop in 2007, the 10% footslope PS treatment typically presented the highest concentration of P, likely due to the conditions described previously (release of iron and calcium- bound P) as well as higher summit groundwater P concentrations. Wider strips like the 10% PS footslope treatment are likely to retain more sediment than thinner strips of perennial vegetation (Tomer, et al., 2007). The PS treatments in contour strips and at the footslope likely trap sediment throughout the catchment for less sediment delivery to the footslope. In post-implementation years, consistent sheet flow may be hard to maintain with sediment deposition (Tomer, et al., 2007), but the lack of uniformity can still effectively trap sediment and associated P (Tomer, et al., 2010).

Infiltration under perennial vegetation has been shown to increase after the second year of growth (Schmitt, et al., 1992). Once established, warm-season perennials such as switchgrass transpire little in early months (Tomer, et al., 2007). The three primary grass species seeded in these PS were also warm-season grasses. Thus, percolation and shallow groundwater recharge prior to increased transpiration in the summer months was possible. Tomer et al. (2007) details this mismatch between plant uptake and nutrient availability as a potential driver for nutrient contamination in shallow groundwater. A study characterizing dissolved P concentration in groundwater at different sites in Iowa indicated that P concentrations in Central Iowa catchments were lower than would be expected from other agricultural sites in Iowa (Burkart, et al., 2004). Kolpin et al. (1996) measured similar P concentrations (<0.01-0.11 mg L^{-1}) from groundwater near the catchments, but the exact locations were unclear.

Shallow groundwater contributes to baseflow at these sites (Schilling and Drobney, 2014). Thus, the supply of P to groundwater may need to be addressed in terms of surface water impact. Surface waters like streams do not typically exhibit anaerobic conditions where P enrichment would be problematic (Correll, 1998). However, P is considered the most important contributing nutrient to eutrophication in freshwater lakes (USEPA, 1990) where dissolved P (mostly orthophosphate) is readily available for algal uptake (Walton, 1971). Studies have shown 0.01-0.02 mg L^{-1} of P were critical levels for noxious aquatic plant growth (Sharpley, et al., 2003; Vollenweider, 1970). Every treatment in this watershed expressed footslope P concentrations at or exceeding this critical range (Table 2.7). Given the export via stream from shallow groundwater baseflow to larger surface water bodies, there is potential that some of the dissolved P will arrive in lakes either in the dissolved state or adsorbed to sediment.

Spikes in P concentration occurring in September 2011 and October 2012 may be partially attributed to the total reactive P measurement since these samples were not filtered in contrast to all other P samples. However, Figure 4 showed for 2015 the relationship between dissolved and total P should be almost a 1:1 ratio. Uncharacteristically large precipitation in 2010 and 2011 and corresponding large runoff events (unpublished data) likely transported P-rich sediment that was trapped by the PS allowing for saturation and release into groundwater the subsequent years.

Overall, the layout of PS within the row crop landscape appeared to be significant for reducing P concentrations in shallow groundwater. Previous studies indicated the importance of controlling P transport at the source (Daniel, et al., 1994). The contour strips slow overland flow in stages, much like terraces, resulting in less transport of P-laden sediment to the footslope where mechanisms allow for easier dissociation into shallow groundwater as dissolved reactive P. The 100% row crop treatment likely exhibits the lowest P concentrations since P-rich sediment was flushed from the system with runoff as opposed to captured, retained, and concentrated at the footslope.

Nutrient Flux

Flux calculations aimed to better quantify nutrient loss produced area-weighted values that may be applied to estimate groundwater nutrient export from any similar watershed. Nitrate-N fluxes (Table 2.8) followed a similar pattern to concentration (Table 2.5) except the fluxes were not all significantly different ($p<0.05$) by treatment. Flux values indicated the presence of PS alone with at least a 10% land conversion in any configuration reduced NO_3-N concentration in groundwater compared to no conversion to PS.

Multiple Iowa studies reported NO_3-N flux lost from conventional row crop land quantified by subsurface drainage measurements with annual NO_3-N flux ranging from 13 to 61 kg N ha^{-1} (Bakhsh, et al., 2005; Drinkwater, et al., 1998; Jaynes, et al., 2001; Li, et al., 2006; Qi, et al., 2011; Tomer, et al., 2003). Based on our concentration measurements and water flux estimations from this study period, approximately 0.37 kg ha^{-1} of NO_3-N is exported from the system within the top 2 meters for the 100% row crop treatment during the 6 month growing season. One possibility for the difference between total leaching values and our estimate is NO_3-N may leach deeper into the soil profile than is accounted for in this study (Foster, et al., 1982). Additionally, sampling provides a snapshot of the current conditions and not a continuous analysis of groundwater nutrient concentrations. It is also important to note the measured flux calculations were limited to the availability of well depth measurements (May-October). Thus, leaching that occurred outside of this time frame was unaccounted for.

Phosphorous fluxes (Table 2.9) significantly varied among treatments similar to P concentration (Table 2.7). The distribution of PS on contours instead of a single footslope position appeared to result in less P export via groundwater from the system. By trapping sediment at multiple locations within the catchment, supersaturation may happen at a lower magnitude, if at all at this point in the study. Also, by holding sediment at higher slope locations, there were less P-rich inputs at the footslope resulting in less saturation (Browne, et al., 2008; Smolders, et al., 2009; Tomer, et al., 2010).

Most work addressing P focused on surface water since P is most likely to be transported adsorbed to suspended particles (Böhlke, 2002). In studies where groundwater samples are taken, NO_3-N was measured, but P concentrations in groundwater were assumed to be insignificant (Heathwaite, et al., 2000). Few studies quantify P flux via groundwater in Central Iowa so it is difficult to compare our estimated fluxes to previous studies.

One early Iowa study reported average annual P losses of 0.003 kg ha^{-1} over a 4-year period (Baker, et al., 1975). However, that study indicated the local subsoils were low in P so the nutrient was likely adsorbed to soil particles (Baker, et al., 1975) resulting in lower P concentrations than might be expected in a more P-rich soil (Daniel, et al., 1994). Direct soil P measurements were not available at this study's catchments, but we may expect higher P flux due to the shallow groundwater tables. Measured flux values range from 2.2 to 6.5 g ha^{-1} (0.002 to 0.007 kg ha^{-1}) for our catchments which may indicate subsoils low in phosphorous.

A recent study in Ohio analyzed dissolved reactive P in groundwater tile lines and found a range in annual fluxes of 0.22 to 0.84 kg ha^{-1} where the highest P concentrations occurred in March, June, and December then the lowest in July, August, and September (King, et al., 2015). In contrast, our data availability was May to October and limited to shallow groundwater flow. Thus, our annual P flux estimation likely underestimated nutrient flux in the groundwater.

Water Balance

Studies reported 19.6 and 24.8 cm of drainage from row crop fields (Lawlor, et al., 2008; Thorp, et al., 2007) which is similar to our calculated infiltration quantity (20.4 cm) for 100% row crop treatments. Utilizing the groundwater flow calculated as a part of the flux equation, we expected shallow flow available for denitrification and PS root interaction to range from 0.63 to 1.61 cm regardless of PS layout for the measured and maximum fluxes, respectively. However, this was limited to the May-October growing season. Without known groundwater nutrient concentrations or depth, runoff depth, or ET during the late fall, winter, and early spring months, it is difficult to estimate the potential full effect of the treatments on nutrient flux in groundwater. Precipitation during the May-October growing season accounted for an average of 74% of the total annual precipitation for the 2007-2014 reporting period.

Conclusions

Significantly lower concentrations of NO_3-N were found in the shallow groundwater footslope wells with the 20% PS treatment compared to any other treatment, and all PS treatments had significantly lower NO_3-N concentrations than the 100% row crop treatment. This likely relates to three factors: denitrification due to shallow water tables, a longer time window for plant uptake of nitrogen given the longer growing season of prairie vegetation compared to row crops, and a reduction in the quantity of fertilizer application due to less row crop production acres within the catchment.

There was no significant difference for P concentrations in shallow footslope groundwater for the 100% row crop and 20% PS treatments. However, the 100% row crop treatment likely exported P-rich sediment with runoff. The 20% PS treatment likely increased sediment deposition within the catchment avoiding high levels of deposited sediment at the footslope where P-releasing conditions were characteristic. Phosphorous flux quantities were highest from the 10% PS at footslope treatment.

Nitrate-N and P flux estimations in shallow groundwater were lower than recorded values from subsurface drainage. This may in part be attributed to the limitation of groundwater table data and nutrient concentration measurements to the May-October growing season in addition to the short-circuited travel time in tile drains. The most significant reductions for both NO_3-N and P flux in shallow groundwater occurred at both the 10% PS in contour and 20% PS in contour treatments.

Future research needs to be conducted to determine the effect of PS in differing locations. One defining characteristic at this site was the shallow groundwater tables that promoted denitrification. At sites with deeper groundwater tables, nitrate would likely leach deeper past prevalent denitrifying conditions, but there could be some treatment effect with PS. Additionally, PS management through mowing, controlled burn, or grazing may affect shallow groundwater nutrient concentrations and should be explored.

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Table 2.1. Catchment characterization (adapted from Gutierrez-Lopez et al., 2014; Zhou et al., 2010)

Catchment	PS Cover	PS Position in Catchment	Area (ha)	Well Elevation MSL (m)	Slope (%)	Sand (%)	Silt (%)	Clay (%)
Basswood 1	10%	Footslope	0.53	294.2	7.5	16.8	57.4	25.8
Basswood 2	10%	Footslope and summit	0.48	294.6	6.6	16.8	57.4	25.8
Basswood 3	20%	Footslope and summit	0.47	293.4	6.4	16.8	57.4	25.8
Basswood 4	20%	Footslope and summit	0.55	290.9	8.2	16.8	57.4	25.8
Basswood 5	10%	Footslope and summit	1.24	288.5	8.9	16.8	57.4	25.8
Basswood 6	100% crop	None	0.84	284.4	10.5	16.8	57.4	25.8
Interim 1	10%	Footslope, side, summit	3.00	290.0	7.7	10.5	66.0	23.5
Interim 2	10%	Footslope	3.19	291.3	6.1	10.5	66.0	23.5
Interim 3	100% crop	None	0.73	289.7	9.3	10.5	66.0	23.5
Orbweaver 1	10%	Footslope	1.18	282.7	10.3	13.0	61.2	25.8
Orbweaver 2	20%	Footslope, side, summit	2.40	295.5	6.7	13.0	61.2	25.8
Orbweaver 3	100% crop	None	1.24	294.1	6.6	13.0	61.2	25.8

Table 2.2. Management practices at the catchments.

Year	Fertilizer Anhydrous (kg N ha^{-1})	Gully Erosion Smoothed	Planting	PS Mowed	Harvest
2007			19-May		9,10-Oct
2008	24-Apr (134.4)		6-May	19,21-May & 25-Aug	22,24-Nov
2009			12-May	25-Jun	20,21-Oct & 2-Nov
2010	10-Apr (184.8)		15-Apr	30-Oct	30,31-Oct
2011			19-May	18,19-Nov	7,8-Oct
2012	27-Mar (156.8)	26-Mar	10-Apr	30-Oct	19,20-Sep
2013			17-May	14-Nov	30-Sep
2014	9,10-Apr (140.1)		6-May		6,7-Nov
2015		4-Apr	6-May	14-Apr§	28,29-Sep
2016	4-Apr (151.2)	2-Apr	26-Apr	11-Apr	3-Oct

§ indicates PS were burned not mowed.

Table 2.3. Average annual variance in groundwater depth at the footslope by treatment ($p<0.05$).

Year	100% Row Crop	10% Footslope Strips	10% Contour Strips	20% Contour Strips
2007	0.16a	0.06ab	0.04ab	0.03b
2008	0.11ab	0.08a	0.02ab	0.02b
2009	0.22a	0.09a	0.07a	0.02a
2010	0.17a	0.05a	0.21a	0.07a
2011	0.52a	0.39a	0.30a	0.21a
2012	0.49a	0.46a	0.70a	0.21a
2013	0.57a	0.41a	0.11a	0.29a
2014	0.09a	0.22a	0.05a	0.02a
2007-2014	0.29a	0.22a	0.19ab	0.11b

Note: Significant differences between treatments are marked with different letters. Horizontal rows show annual comparisons while the last row shows a comparison across the whole time period.

Table 2.4. Average annual summit NO_3-N concentration in mg L^{-1} ($p<0.05$).

Year	100% Row Crop	10% Footslope Strips	10% Contour Strips	20% Contour Strips	Number Censored
2007	0.85a	0.37a	0.53a	0.47a	29
2008	2.16a	1.64a	1.46a	1.35a	3
2009	2.17a	2.04a	1.33a	1.64a	4
2010	4.62a	3.42a	3.41ab	1.98b	3
2011	4.76a	3.17b	3.09b	2.79b	0
2012	4.74ab	3.59a	3.13b	2.43ab	0
2013	5.96ab	6.20a	3.19bc	2.23c	0
2014	7.43ab	7.17a	4.42bc	2.86c	0
2015	5.85b	7.88a	5.03b	2.92c	0
2016	5.74a	6.03a	4.40ab	2.65b	0
2007-2016	4.43a	4.15a	3.00b	2.13c	39

Note: Significant differences between treatments are marked with different letters. Horizontal rows show annual comparisons while the last row shows a comparison across the whole time period. Number censored indicates the number of samples that were below the lowest standard for analysis.

Table 2.5. Average annual footslope NO_3-N concentration by treatment in mg L^{-1} ($p<0.05$).

Year	100% Row Crop	10% Footslope Cover	10% Contour Strips	20% Contour Strips	Number Censored
2007	0.59a	0.54ab	0.61a	0.26b	29
2008	3.84a	1.38b	0.51b	0.51b	24
2009	2.09a	0.59b	0.63ab	0.29b	30
2010	3.99a	0.97bc	0.88ab	0.44c	22
2011	3.30a	1.50b	1.39b	0.87c	1
2012	4.97a	1.56bc	1.73b	1.11c	0
2013	7.17a	1.14b	1.32b	1.40b	1
2014	8.78a	1.07b	1.66b	1.43b	0
2015	6.13a	0.92c	2.24b	0.90c	1
2016	5.87a	2.17b	3.09b	1.54c	0
2007-2016	4.67a	1.19c	1.40b	0.88d	108

Note: Significant differences between treatments are marked with different letters. Horizontal rows show annual comparisons while the last line shows a comparison across the whole time period. Number censored indicates the number of samples that were below the lowest standard for analysis.

Table 2.6.Average annual summit P concentration in mg L^{-1} ($p<0.05$).

Year	100% Row Crop	10% Footslope Cover	10% Contour Strips	20% Contour Strips	Number Censored
2007	0.01ab	0.01a	0.00b	0.01ab	25
2008	0.01b	0.03a	0.02b	0.02ab	0
2009	0.02a	0.03a	0.03a	0.04a	0
2010	0.01a	0.02a	0.07a	0.02a	2
2011	0.04a	0.09a	0.03a	0.11a	31
2012	0.00a	0.05a	0.04a	0.02a	21
2013	0.02b	0.13a	0.03b	0.03b	18
2014	0.02b	0.05a	0.06ab	0.02b	17
2015	0.03a	0.03a	0.02a	0.03a	17
2016	0.03ab	0.07a	0.03ab	0.02b	2
2007-2016	0.02b	0.05a	0.03b	0.03b	133

Note: Significant differences between treatments are marked with different letters. Horizontal rows show annual comparisons while the last line shows a comparison across the whole time period. Number censored indicates the number of samples that were below the lowest standard for analysis.

Table 2.7. Average annual footslope P concentration in mg L^{-1} ($p<0.05$).

Year	100% Row Crop	10% Footslope Cover	10% Contour Strips	20% Contour Strips	Number Censored
2007	0.008b	0.073a	0.026b	0.012b	6
2008	0.04ab	0.090a	0.021b	0.012b	0
2009	0.013a	0.077a	0.027a	0.021a	3
2010	0.028a	0.032a	0.077a	0.048a	0
2011	0.032b	0.246a	0.071b	0.043b	28
2012	0.011b	0.196a	0.038b	0.020b	27
2013	0.027b	0.095a	0.044b	0.014b	29
2014	0.017b	0.106a	0.046b	0.025b	14
2015	0.014c	0.142a	0.071b	0.021bc	16
2016	0.012b	0.076a	0.094a	0.052a	10
2007-2016	0.020c	0.113a	0.052b	0.027bc	133

Note: Significant differences between treatments are marked with different letters. Horizontal rows show annual comparisons while the last line shows comparisons across the whole time period. Number censored indicates the number of samples that were below the lowest standard for analysis

Table 2.8. Estimated 6-month growing season NO_3-N flux per treatment in kg ha^{-1} yr^{-1} ($p<0.05$).

Year	100% Rowcrop	10% Footslope Cover	10% Contour Strips	20% Contour Strips
2007	0.01a	0.01a	0.02a	0.02a
2008	0.52a	0.17b	0.07b	0.07b
2009	0.10a	0.03b	0.02b	0.03ab
2010	0.49a	0.13b	0.10b	0.06b
2011	0.36a	0.14ab	0.10b	0.02ab
2012	0.15a	0.05a	0.02a	0.03a
2013	0.70a	0.02b	0.08b	0.11b
2014	0.61a	0.05b	0.12b	0.08b
2007-2014	0.37a	0.08b	0.07b	0.05b

Note: Significant differences between treatments are marked with different letters. Horizontal rows show annual while the last line shows comparisons across the whole time period. Number censored indicates the number of samples that were below the lowest standard for analysis

Table 2.9. Estimated 6-month growing season P flux per treatment in g ha^{-1} yr^{-1} ($p<0.05$).

Year	100% Rowcrop	10% Footslope Cover	10% Contour Strips	20% Contour Strips
2007	2.39a	4.37a	0.66a	0.80a
2008	10.22a	14.77a	1.84b	1.93b
2009	0.87b	3.40a	0.80b	1.16b
2010	3.71a	2.34a	9.02a	2.95a
2011	2.40b	13.76a	2.09b	0.80b
2012	0.23b	4.87a	0.27b	0.55b
2013	1.14a	0.91a	1.37a	1.32a
2014	1.04b	7.48a	1.69b	2.18b
2007-2014	2.75b	6.49a	2.22b	1.46b

Note: Significant differences between treatments are marked with different letters. Horizontal rows show yearly comparisons while the last line shows comparisons across the whole time period.

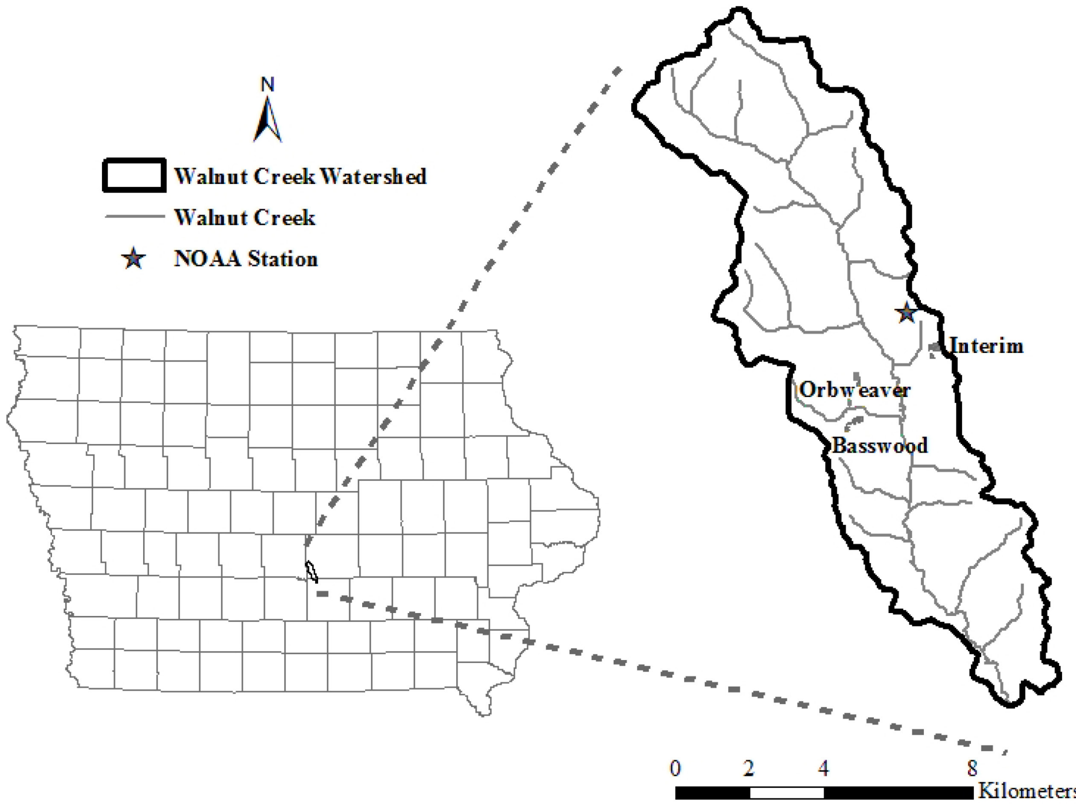

Figure 2.1. Location of study catchments within the Walnut Creek Watershed in Jasper County, Iowa (adapted from Zhou et al., 2010).

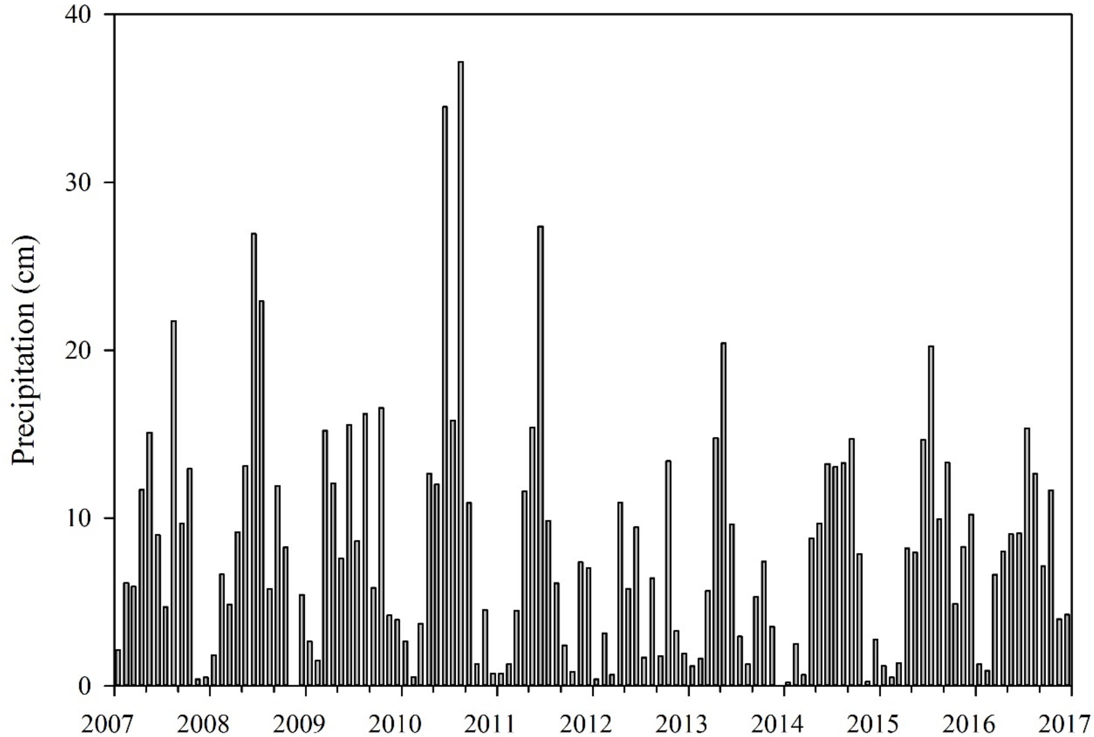

Figure 2.2. Monthly precipitation from MesoWest Station NSWI4.

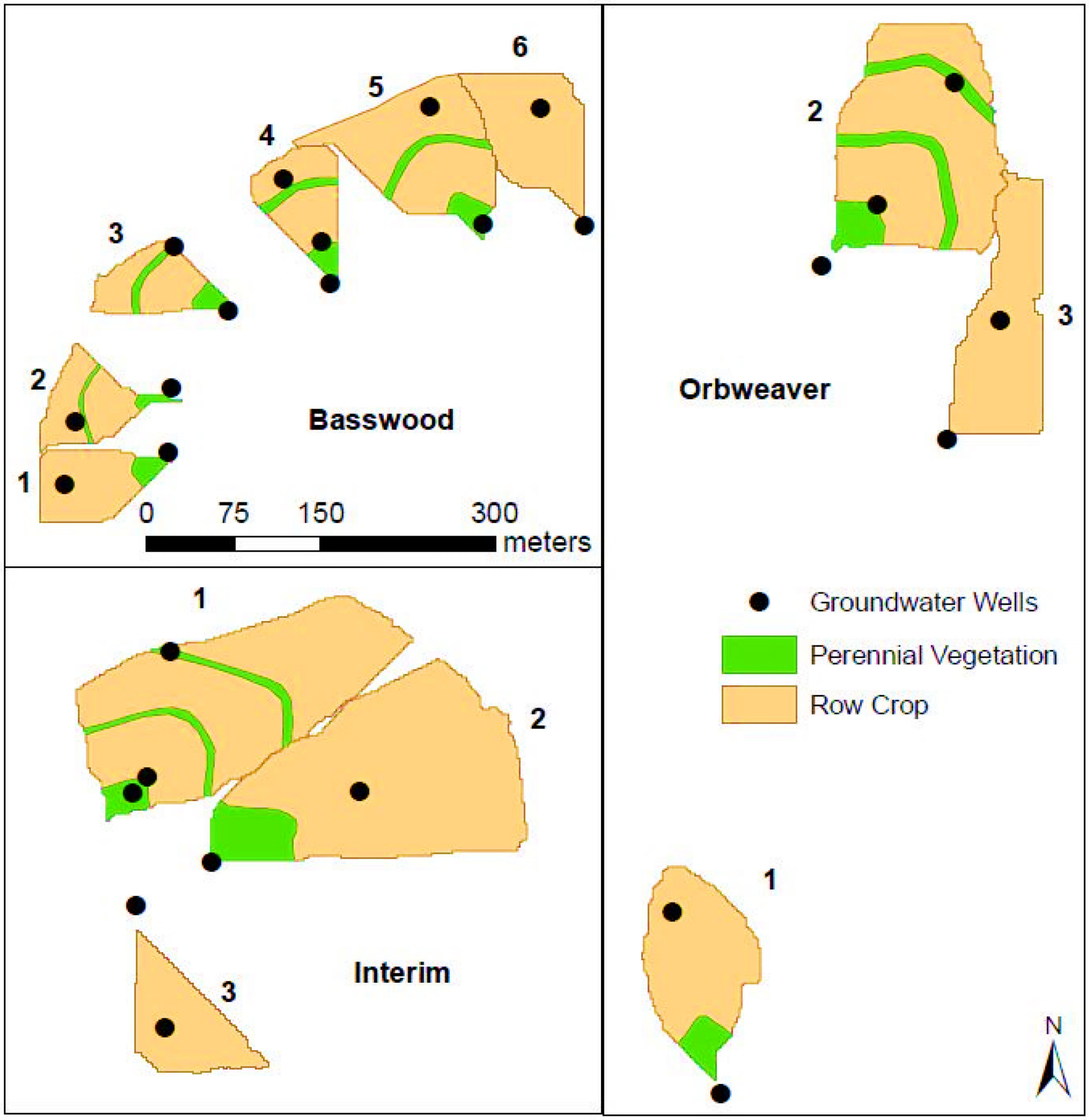

Figure 2.3. Catchment delineation with treatment layout (adapted from Zhou et al., 2010).

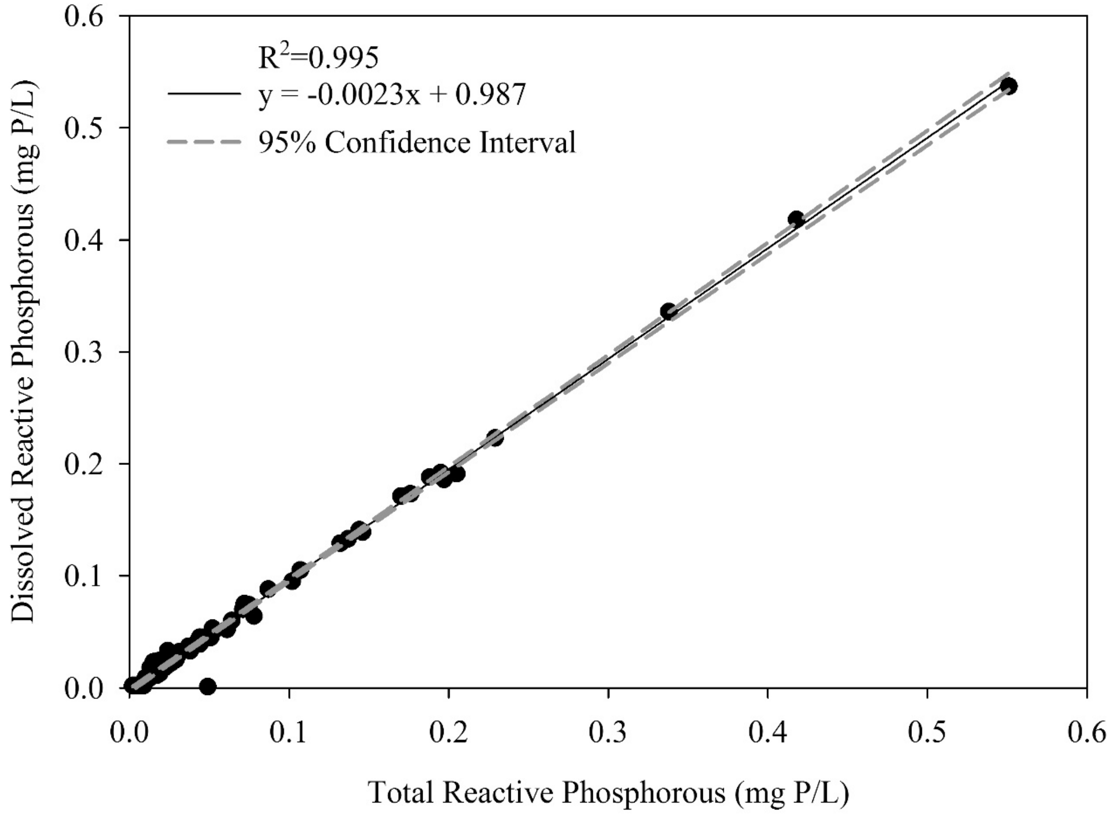

Figure 2.4. Regression comparing total reactive P (unfiltered sample) to dissolved reactive P (filtered). The nearly 1:1 slope indicates filtering has little effect on these measurements.

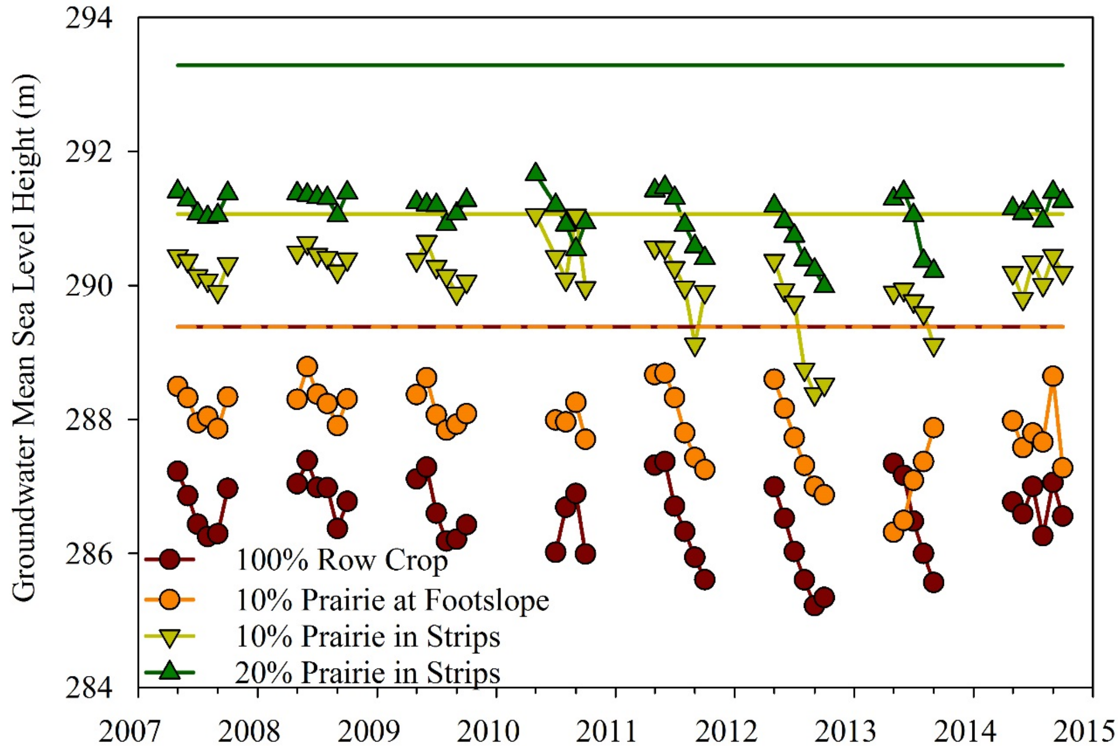

Figure 2.5. Monthly fluctuation in groundwater levels averaged monthly by treatment. Lines of corresponding color indicate soil surface level at mean sea level averaged by treatment.

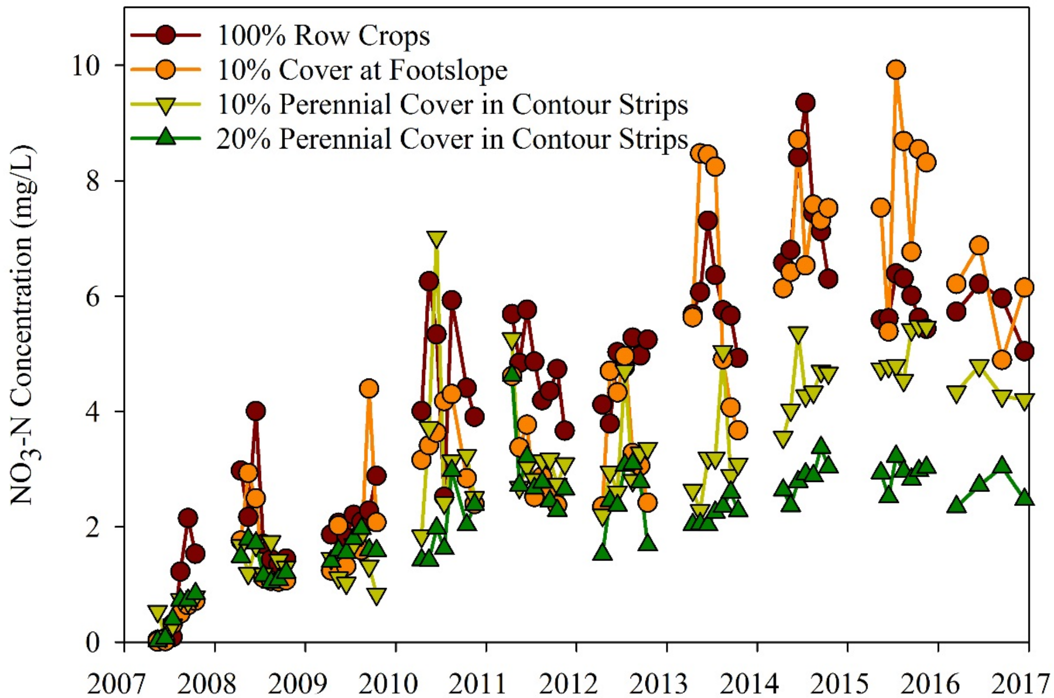

Figure 2.6. Monthly NO_3-N concentration in summit shallow groundwater.

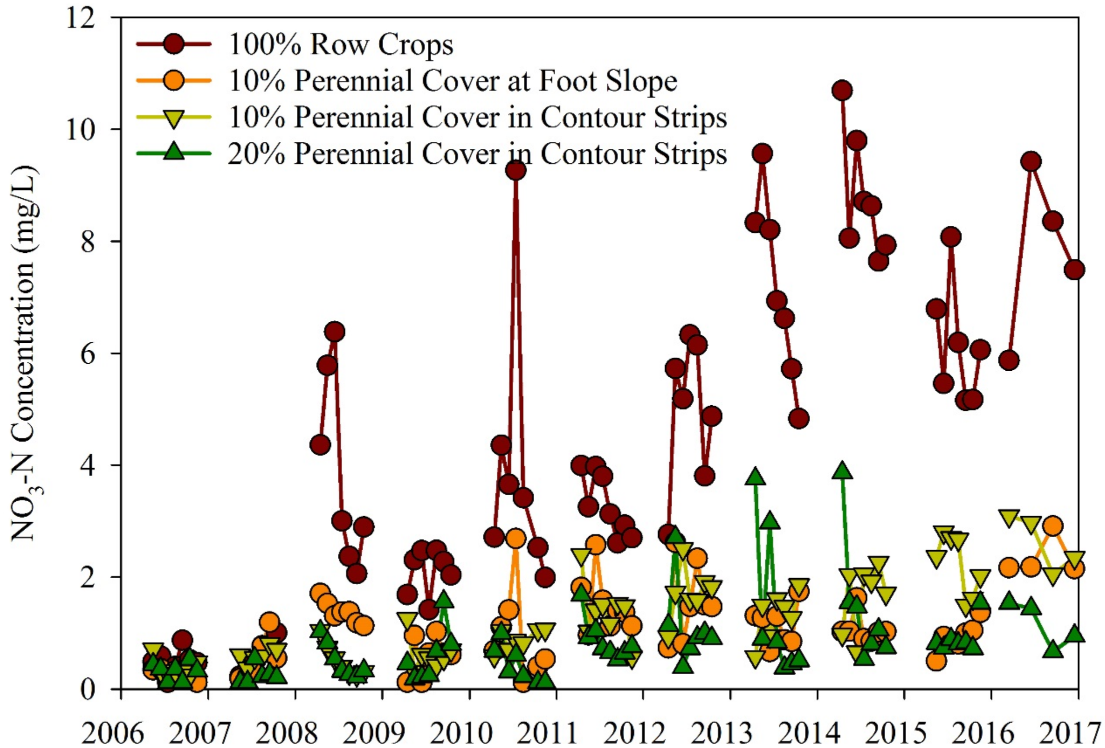

Figure 2.7.Monthly NO_3-N concentration in footslope shallow groundwater.

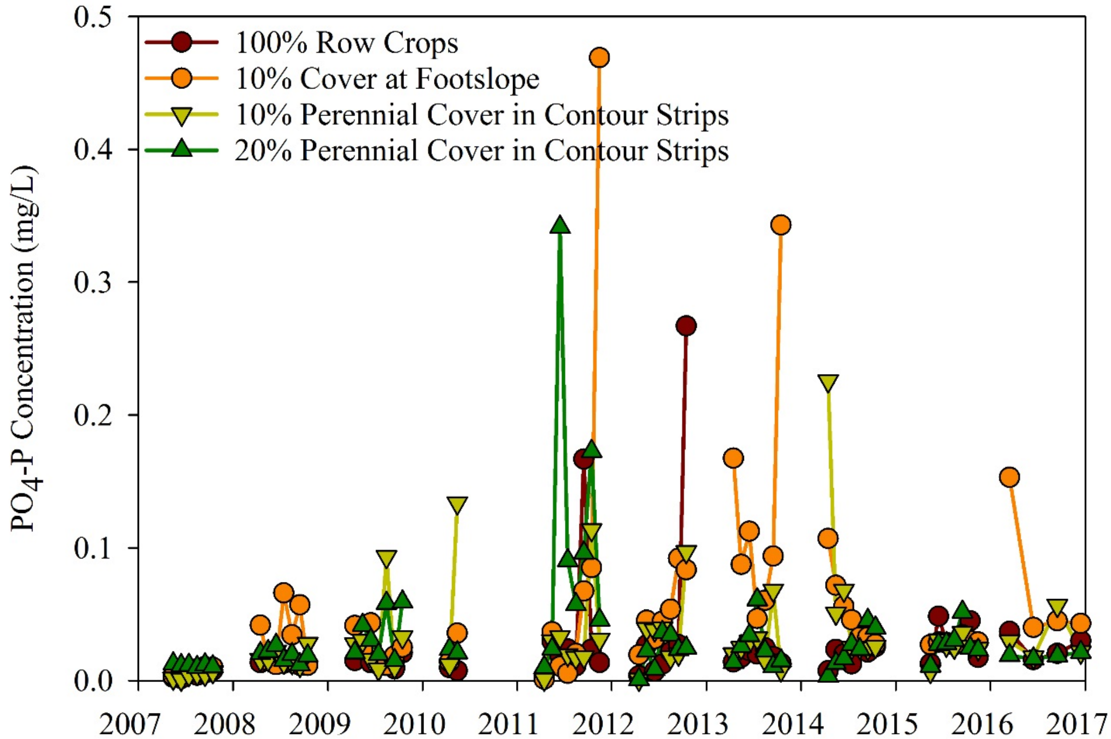

Figure 2.8.Monthly P concentration in summit shallow groundwater.

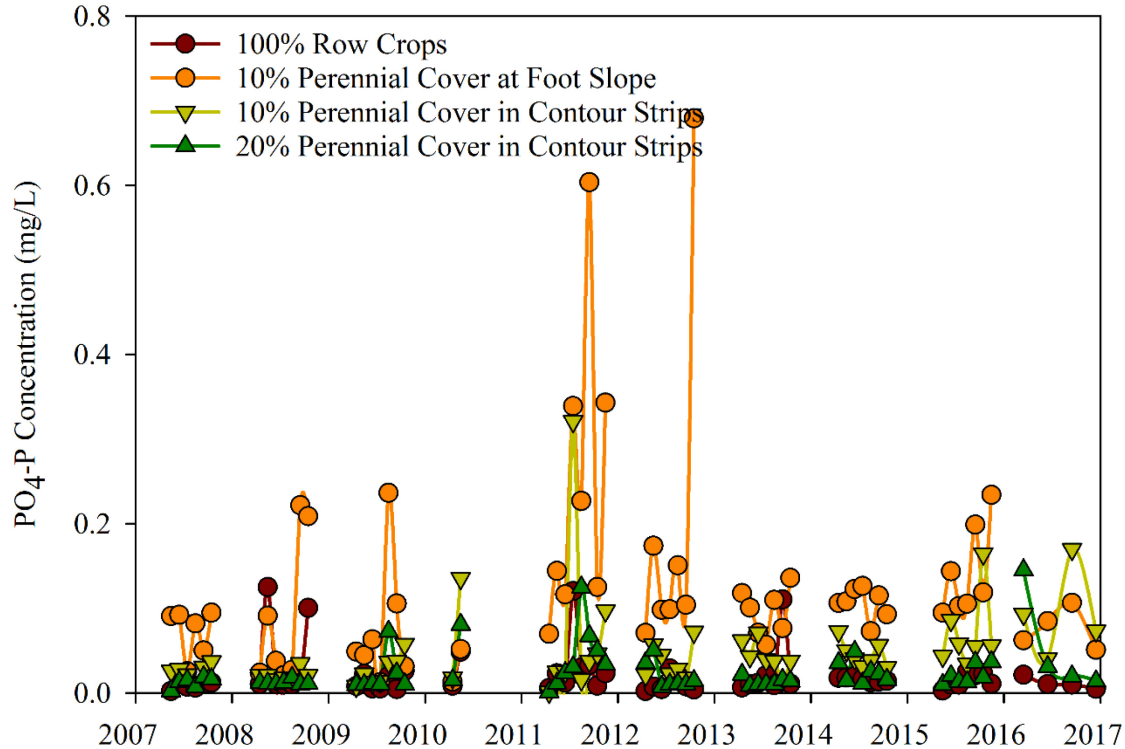

Figure 2.9. Monthly P concentration in footslope shallow groundwater.

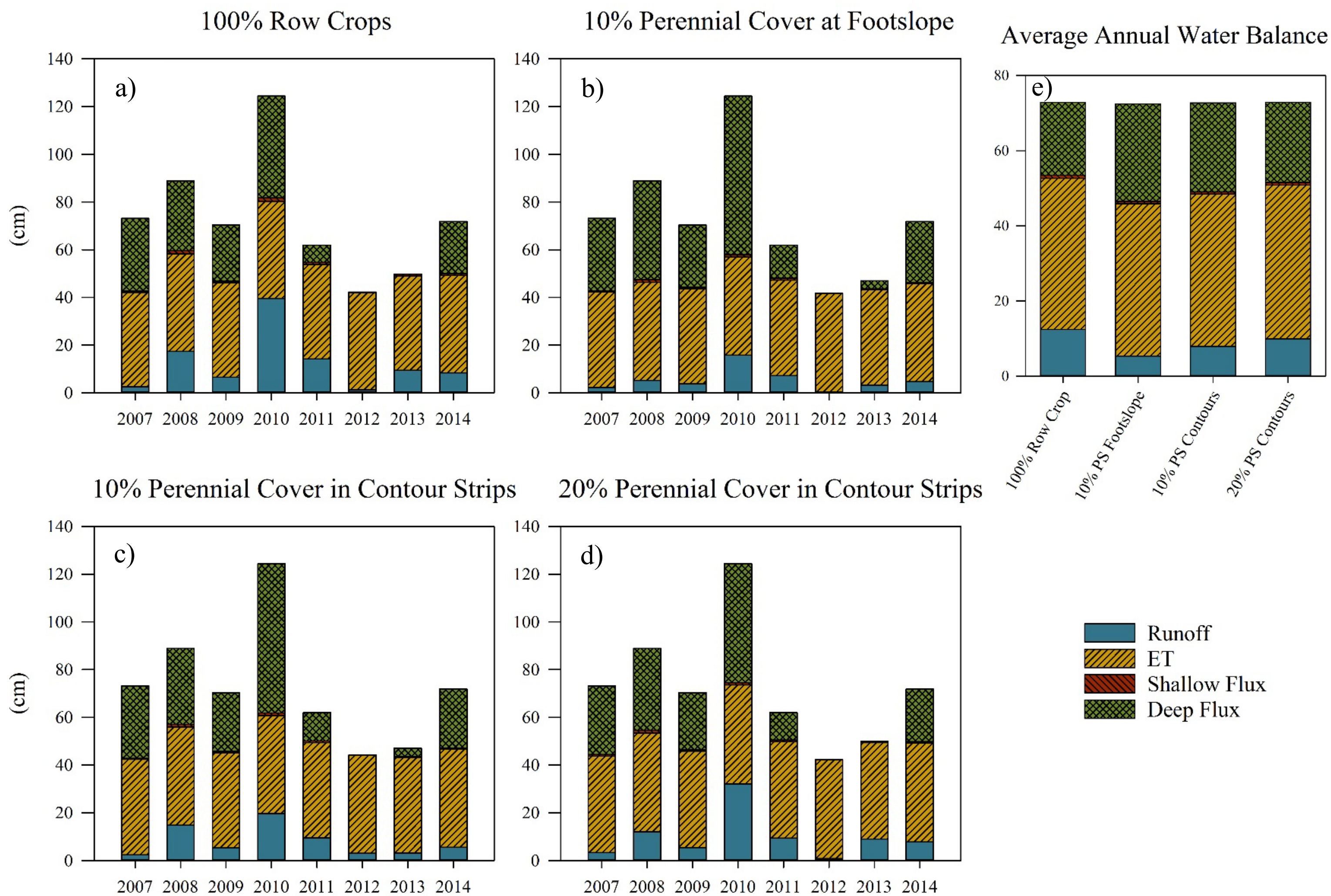

Figure 2.10. Water balance by treatment and annual average for 6 month growing season.

CHAPTER 3. CHRONOSEQUENCE OF SOIL HEALTH PARAMETERS FOLLOWING CONVERSION FROM ROW CROP TO PRAIRE

A paper to be modified for submission to *Journal of Soil and Water Conservation*

Bethany A. Brittenham[2], Randy K. Kolka, Dan S. Andersen, Mark D. Tomer, Matthew J. Helmers

Abstract

Conversion of row crop to prairie has been shown to modify a myriad of soil physical and chemical properties. Thus, the objective of this study was to quantify soil property changes following conversion from row crop to prairie. This study included data from 6 sites with restored prairie vegetation ranging in age from 2, 10, 25, and 37 years and row crop fields for comparison. Due to the importance of soil genesis, particle size distribution, and precipitation, the 37 year chronosequence analysis of soil properties was isolated to 3 Central Iowa sites with similar soil series. Results were also provided from 3 sites in differing regions of Iowa at 2 years post-conversion. Properties reviewed were total carbon, total nitrogen, pH, bulk density, infiltration, soil particle size, aggregate size distribution with total carbon and total nitrogen content, particulate organic matter as total carbon and total nitrogen content, and mineral associated organic matter as total carbon and total nitrogen content.

Soil properties for the 0-5 cm depth varied significantly across the chronosequence. The total carbon to total nitrogen ratio and pH increased significantly following conversion from row crop to prairie while bulk density decreased significantly following conversion to

[2] Primary author and researcher

prairie. Quantity of large macroaggregates significantly increased with time in prairie as well as aggregate fraction carbon and nitrogen content. Total carbon in the fine particulate organic matter pool increased significantly with time as well as the total carbon to total nitrogen ratio for particulate organic matter and mineral associated organic matter. Whole soil total carbon appeared to increase significantly then level off with time. Whole soil total nitrogen, mineral associated organic matter total carbon content, and aggregate fraction total carbon to total nitrogen ratio did not express a significant change with time. Three 2 year post-conversion sites with corresponding row crop fields show mixed results for the aforementioned soil properties as well as corresponding infiltration measurements. Thus, future resampling will be required to assess a clear trend.

Results from this study may be utilized to develop expected trends in soil properties following conversion from row crop to prairie for the chronosequence region. Future measurements may be compared back to current benchmark analyses for all sample regions.

Introduction

Conversion of tallgrass prairie to production agriculture has become so extensive in Iowa that less than 1% of the historical 12.5 million hectares remains (Samson and Knopf, 1994; Smith, 1990). This change in land cover from perennial vegetation to row crop disrupts nutrient cycling and organic matter turnover as well as reduces soil aggregate size (Buyanovsky, et al., 1987; Davidson and Ackerman, 1993; Dinnes, et al., 2002; Freibauer, et al., 2004; Jenny, 1941; Tisdall and Oades, 1982). A result of land conversion, approximately 50% of the soil organic carbon in the top 20 cm of Central Iowa soil was lost due to erosion and tillage (Donigian, et al., 1995). An additional Iowa study on soil organic carbon content by erosion class indicated severely eroded soils contained approximately half the quantity of soil organic carbon of slightly eroded soils (Kimble, et al., 1999).

Following conversion from row crop to perennial vegetation, previous studies have measured carbon accumulation in perennial vegetation biomass as well as an increase in soil organic carbon (Anderson-Teixeira, et al., 2009; Freibauer, et al., 2004; Gebhart, et al., 1994; Insam and Domsch, 1988; Knops and Tilman, 2000; Post and Kwon, 2000). Rate of carbon accumulation has been shown to vary by location, climate, soil texture, and vegetation composition (Christensen, 1996; Knops and Tilman, 2000) similar to Jenny's (1941) 'factors of soil formation': climate, biota, relief, parent material, and time. Soil carbon saturation theory (Six, et al., 2002) presents four soil carbon pools: silt and clay associated, physical protection within aggregates, biochemical protection within complex compounds (i.e. lignin), and an unprotected carbon pool.

Results from the Knops and Tilman (2000) study suggest the rate of carbon accumulation is controlled by nitrogen accumulation. Likely nitrogen sources for perennial vegetation include atmospheric deposition, microbial fixation, and redistribution within the soil profile (Knops and Tilman, 2000). Deposition alone may account for nitrogen accumulation within the soil (Anderson and Downing, 2006; Howarth, et al., 2002a). For prairie vegetation near row crop sites, leached nitrate-nitrogen in groundwater may artificially bolster plant communities and increase carbon stocks within prairie land cover while simultaneously accumulating nitrogen (Schipper, et al., 2004; Springob and Kirchmann, 2003).

A structural characteristic, aggregate size distribution is known to vary by land cover and season (Harris, et al., 1966; Mulla, et al., 1992). In general, aggregate formation may occur primarily in spring due to moisture availability and soil organic matter then degrades throughout the year if the soil surface is bare (Harris, et al., 1966). Aggregate size distribution may help assess soil erodibility and aid in selecting management practices to prevent soil loss and increase infiltration (Bharati, et al., 2002; Kemper and Rosenau, 1986; Le Bissonnais, 1996). Infiltration has been shown to increase 2 years post conversion from row crop to perennial vegetation (Schmitt, et al., 1992).

Carbon and nitrogen storage within the soil typically varies by aggregate size with microaggregates (<0.21 μm) containing a lower organic content than macroaggregate (>0.21 μm) fractions (Cambardella and Elliott, 1993; Dormaar, 1983). Destruction of the protective aggregates releases particulate organic matter into a labile pool spurring organic matter mineralization and carbon release (Six, et al., 2002). Thus, the destruction of nutrient-rich macroaggregates and conversion to nutrient-poor microaggregates may reduce soil capacity for nutrient cycling over time as carbon content wanes (Elliott, 1986).

Particulate organic matter (POM) represented the balance of primary productivity and decomposition. It served as a sensitive measure of change and ecosystem function (Burke, et al., 1989; Cambardella and Elliott, 1992). Studies indicated POM quantity changed with inputs and management practices more quickly than the total carbon pool (Dalal and Mayer, 1987; Hassink, 1997). Thus we expected a significant increase in POM content by prairie age. In contrast, silt and clay associated carbon and nitrogen form finite, protected pools that may not express significant differences with prairie age if the pool is saturated.

Chronosequences aim to detail expected changes in measurable soil properties following a modification of land management practices. Comparable data sets may include samples at the same sites over many years or multiple sites with similar basic properties like soil genesis and slope. With consistent management, chronosequences provide the opportunity to determine if soil properties have stabilized to relative equilibrium (Stewart, et al., 2007).

Reviews indicated research of land change effects on soil properties has been biased toward tropical environments and forest to grassland conversion (Post and Kwon, 2000). Thus, the reversion of row crop to prairie provides valuable insight on soil property alteration 10, 25, and 37 years post conversion with a row crop field for reference. Added comparisons at 3 distinct, additional sites (collectively referred to as Phase II) evaluated soil properties from sites 2 years post-conversion to prairie vegetation paired with row crop treatments within the same field. Phase II sites provided a baseline for future work. The objectives of this study were to (1) determine if the chronosequence sites have reached relative equilibrium for multiple parameters 37 years post-conversion, (2) assess potential of land conversion to accumulate carbon, and (3) compare baseline soil properties at paired comparison sites.

Materials and Methods

Site Descriptions

This study combined data from 6 sites at 5 distinct locations within Iowa (Figure 3.1). From west to east, sites were as follows: Armstrong (ARM; 41°18'N, 95°10'W), Neal Smith National Wildlife Refuge (NSNWR; 41˚33'N; 93˚14'W), Rhodes (RHO; 41°53'N, 93°12'W), Jacob Krumm Nature Preserve (KRU; 41°42'N, 92°46'W), and Eastern Iowa Airport (EIA; 41°53'N, 91°43'W). ARM, NSNWR, RHO, and KRU resided within the Pre-Illinoian Southern Iowa Drift Plain while EIA was in the Pre-Illinoian Iowan Surface (Prior, 1991). The Southern Iowa Drift Plain is characterized by rolling hills with abundant groundwater and streams where soils are primarily mollisols and alfisols with some entisols (NRCS, 2006). The

Iowan Surface is known for gently rolling long slopes and glacial deposits dominated by mollisols and alfisols (NRCS, 2006).

ARM, RHO, and EIA sites were 100% row crop prior to the addition of prairie strips (PS) planted in 2014. These sites had a corresponding controlled pair where PS were not incorporated and are collectively referred to as Phase II. NSNWR contained two sites for this study. One was converted from brome grass to row crop with PS in 2007 and is hereto referred to as Interim 1 (IN1). The other NSNWR site was converted from brome grass directly to prairie in 1992 and will be referred to as Interim 4 (IN4) for this paper. Restoration of KRU to prairie from row crop agriculture began in 1980. Thus, Phase II and prairie restoration sites provided a chronosequence of 2, 10, 25, and 37 years for observed changes in soil properties following the conversion from row crop to prairie.

Sample Locations

Sampling locations were developed from SSURGO data (USDA-NRCS, 2004; 2005a; b; c). For each site, 3 samples per soil type were randomly sited within prairie vegetation and labeled ‘PS’. For Phase II sites, a ‘row crop’ point was defined as 3 m upslope from the edge of the PS where the corresponding sample was taken forming a PS and row crop pair (Table 3.1, Figures 3.2-3.5). For the Phase II sites, an additional 3 sample sites per soil type (if available) were randomly assigned in the control field and labeled ‘control’ (Table 3.1, Figures 3.2-3.4). Samples for the KRU site were taken in soil series corresponding to those available at IN1 and IN4 (Figures 3.5-3.7).

Soil Sample Techniques

Soil cores were taken in the fall of 2015 to assess general soil properties at the ARM, RHO, and EIA sites according to the positions described previously. A Giddings brand coring machine (Windsor, CO) was used to extract 4 cm by 120 cm cores that were stored at 4˚C prior to processing. Cores were then cut into depth increments at 5, 10, 20, 40, 60, 80, and 100 cm and air dried. A 10 g subsample was removed and oven dried at 105 ˚C in a Humboldt Batch Oven (Eling, IL) for 24 hours to determine the percent water content for bulk density calculations. The remaining sample was then ground to pass through a 0.25 mm sieve and stored in zip top bags awaiting analysis. A similar procedure was followed for NSNWR and KRU sites, but samples were taken with a hand probe in fall 2016 to a depth of 15 cm and cut at 5 and 15 cm depth increments. Additional hand probe samples were taken at Phase II sites in fall 2016, dried, and sieved awaiting pH analysis.

Soil Chemical Properties

Analyses were run on the top 3 depths (0-5, 5-10, and 10-20 cm) for the Phase II sites as is typical for chronosequences (Breuer, et al., 2006; Burke, et al., 1989). Chronosequence sites were analyzed for the 0-5 cm depth and the remaining 5-15 cm depth samples archived. Total nitrogen (TN) and total carbon (TC) percentages were quantified by combustion with a LECO 628 Series (Saint Joseph, MI). The second set of 0-5 cm depth samples taken in fall 2016 were analyzed for pH via water extraction with a Fisher Scientific Accumet Basic AB15 Plus pH meter (Agawam, MA).

Soil Physical Properties

Bulk density was determined with the measured volume and calculated oven-dry mass of the sample with Equation 1 (Blake and Hartge, 1986)

$$\rho_b = \left(\frac{m_s}{V_c}\right) = \frac{m_t - m_w}{\pi * \left(\frac{d}{2}\right)^2 * l} \tag{1}$$

where ρ_b is the bulk density (g cm^{-3}), m_s is the mass of the soil particles (g), V_c is the core volume (cm^3), m_t is the total sample mass prior to drying (g), m_w is the mass of water lost by oven drying (g), d is the core diameter (cm), and *l* is the core length (cm). Particle size analysis from the fall 2016 samples was determined by the hydrometer method (Blake and Hartge, 1986).

Infiltration

Cornell Sprinkle Infiltrometers (Ithaca, NY) were utilized at PS and row crop points to assess runoff rate, infiltration rate, and field-saturated infiltration (van Es and Schindelbeck, 2015). Use was limited to the Phase II sites (ARM, RHO, and EIA) given time constraints. ARM and EIA measurements were made summer 2016 while RHO infiltration was done summer 2017. Thus, measurements reflect infiltration at sites 2, 2, and 3 years post-conversion to prairie. It was expected that steady-state infiltration would not be measurable via runoff quantification given the increased infiltration under long-term perennial plants (Bharati, et al., 2002) and maximum rainfall rates for Cornell Sprinkle Infiltrometers (van Es and Schindelbeck, 2015). A constant rainfall rate was simulated by the infiltrometer while runoff volume was recorded at 3 minute intervals following initial runoff. Once steady runoff volume conditions were measured for 3 intervals (within 10 mL), infiltration measurements concluded for that data point.

Data analysis began by calculating rainfall and runoff rates (Equation 2)

$$r = \frac{(h_1 - h_2)}{t_f} \tag{2}$$

where r was the simulated rainfall rate (cm min^{-1}), h_1 was the water height (cm) at the beginning of the time interval, h_2 was the water height (cm) at the end of the interval, and t_f was the time for the difference in height to occur (min). The time interval runoff rate was then calculated from the runoff volume (Equation 3)

$$ro_t = \frac{V_t}{A * t} \tag{3}$$

where ro_t was the runoff rate (cm min^{-1}), V_t was the measured volume of water that ran off the soil surface (cm^3), A was the area of the ring (457.30 cm^2), and t was the time interval (min). The infiltration rate was simply the difference in rainfall and runoff rates (Equation 4).

$$i_t = r - ro_t \tag{4}$$

Smoothing runoff and rainfall rates across 3 measurement intervals was suggested since steady rainfall simulation rates may be hard to maintain in field conditions (Schindelbeck, personal communication, 2016). Thus, initial and final values were maintained while intermediate values were averaged with the previous and subsequent measurement.

Field-saturated infiltration (Equation 5) was compared among treatments (Reynolds and Elrick, 1990; van Es and Schindelbeck, 2015)

$$i_{fs} = i_t * 0.80 \tag{5}$$

where i_{fs} was the field-saturated infiltration and i_t was the infiltration rate. The 0.80 factor was necessary to account for three-dimensional flow at the base of the ring in loamy soil with a 7.5 cm insertion depth (Reynolds and Elrick, 1990).

Aggregate Size Distribution

Samples were taken from late October to early December in 2016 to acquire a snapshot of the post-harvest soil conditions and assure each site experienced similar weather patterns (Cambardella, personal communication, 2016; Mulla, et al., 1992). For this analysis, samples were composites of 15 subsamples taken with a 2.54 cm diameter push probe. At each paired PS and row crop point, 3 different sample classifications were taken: within-strip, within-row, and between-row. Within-strip samples were taken in the PS avoiding patches devoid of vegetation. Within-row samples were taken between row crop stubble in the crop rows. Between-row samples were taken in the middle of the inter-row spaces within row crop. Samples were divided into two depths (0-5 and 5-15 cm) and bagged separately prior to storage at 4˚C. Care was taken to avoid track rows influenced by mechanical compaction.

Preprocessing began by passing samples through an 8 mm sieve at field-moist conditions and breaking along natural fractures (Ontl, et al., 2015). Gravel greater than 8 mm was extracted and dried to determine mass. Roots greater than 1 cm in length were removed. Samples were then air-dried to a constant mass, hand-stirring daily. A 10 g subsample was extracted and dried at 105 ˚C to determine air-dried moisture content. Air-dried samples were stored in zip top bags while awaiting further analysis.

In preparation for wet-sieving, field capacity for each site was determined based on soil particle size, percent organic matter, and bulk density (Saxton and Rawls, 2006). In a plastic petri dish, 100 g of air-dried sample was capillary wetted using DI water and filter paper to field capacity plus 5%, taped shut, and stored at 4˚C overnight (Márquez, et al., 2004; Six, et al., 1998). All the 0-5 cm samples and 10% of the 5-15 cm samples were wet-sieved.

The following day, moist aggregates were spread on a nest of sieves with 2.00, 1.00, and 0.21 mm openings (Ontl, et al., 2015) and wet-sieved similar to the Yoder wet-sieving method with a 10 minute cycle, 4 cm stroke length, and a frequency of 30 cycles min^{-1} (Mikha and Rice, 2004; Yoder, 1936). Care was taken that aggregates on the top sieve were covered with water at the top of the upstroke and water did not run over the outer edge of the sieve at the bottom of the down stroke (Nimmo and Perkins, 2002). Aggregates and sand retained on each sieve were then backwashed into pre-weighed tins and oven dried at 60˚C for 24-48 hours or until dry. By definition, macroaggregates are aggregate fractions greater than 0.21 mm. Particles that were not retained on a sieve at the end of the cycle (microaggregates) were discarded with the sieving water after each run. Approximately 10 g of each macroaggregate fraction was ground with a mortar and pestle prior to combustion analysis for TC and TN with a LECO TruSpec CN (Saint Joseph, MI). The remaining macroaggregate fractions were archived in coin envelopes.

Whole-Soil Particulate Organic Matter

Particulate organic matter (POM) separations were done on whole-soil samples for the 0-5 cm depth. Approximately 30 g of air-dried sample was sieved through a 2.0 mm sieve where organic matter and gravel greater than 2.0 mm was removed by hand. A 30 mL solution of 5% w v^{-1} sodium hexametaphosphate was used to disperse 10 g of the sieved sample overnight on a reciprocating shaker (Cambardella, et al., 2001; Ontl, et al., 2015). The dispersed sample was then rinsed through 0.50 and 0.053 mm sieves until the distilled water ran clear (Ontl, et al., 2015).

The 2.0-0.50 mm fraction was designated coarse POM and sand while the 0.50-0.053 fraction was designated fine POM and sand. The fraction passing through the 0.053 sieve was mineral associated organic matter (MAOM). All three fractions were oven dried at 60˚C and stored in coin envelopes prior to combustion analysis for TN and TC with a LECO TruSpec CN (Saint Joseph, MI). The fine POM (0.053-0.50 mm) and MAOM (<0.053 mm) fractions were analyzed for TC and TN. Coarse POM (0.50-2.0 mm) TC and TN quantification was intended by subtraction of the fine POM and MAOM from the whole-soil TC and TN content.

Statistical Analysis

Analyses of TN, TC, TC:TN ratio, pH, and bulk density within the 0-5 cm soil samples were done with a general linear model (SAS, 2012) for the chronosequence sites (IN1, IN4, and KUR). Soil properties were analyzed based on time since conversion to prairie from row crop. Thus row crop and prairie samples in both Phase II and restoration sites were assigned the corresponding number of years since conversion to prairie was initiated (0, 2, 10, 25, and 37). Phase II (ARM, EIA, and RHO) soil properties were analyzed with paired t-Tests to account for the paired PS and row crop design. Phase II sites were run separately from the chronosequence sites due to variability likely caused by different climate, biota, relief, parent material, and soil age (Jenny, 1941).

The Shapiro-Wilk statistic indicated the infiltration data was not normally distributed. Thus, the Wilcoxon Signed Rank Test was used to determine if the median difference between treatment pairs was significant for field saturated infiltration.

For aggregate and POM fractions, within-row and between-row samples were combined representing the crop treatment as a whole for chronosequence and paired sites. Use of the Wilcoxon Signed Rank Test between within-row and between-row pairs from the same sample location indicated no significant difference in carbon or nitrogen distribution between aggregate and POM carbon and nitrogen content for these sample pairs.

Aggregate size distribution and POM analyses utilized the general linear model for the chronosequence samples. Analysis was run on aggregate retention per sieve on the basis of time since conversion to prairie. Chronosequence time was categorized by year as 0, 10, 25, and 37 years with the 0-year treatment designated for samples taken within row crop. Soil type was not significant in the model ($p>0.300$) and was pooled with the random error. For the Phase II samples, aggregate size distribution, POM, and the corresponding quantities of carbon and nitrogen within each fraction were analyzed with paired t-Tests within each field. The benchmark value for significant difference was $p<0.10$ for all analyses.

Results

Total Nitrogen

Phase II paired comparisons by field indicated no significant difference in TN content within the top 5 cm of soil from EIA and RHO sites between row crop, PS, and control treatments (Table 3.2). At the ARM site, TN content was significantly different between the control field samples and both the row crop and PS samples. The row crop and PS TN samples were not significantly different for the ARM site.

Sites utilized for the chronosequence comparison (IN1, IN4, and KRU) showed no significant difference among the 0, 10, 25, and 37-year prairie treatments for TN (Table 3.3). The difference in TN from the row crop and 37-year prairie indicated an average yearly increase in soil TN of 0.01 g N m^{-2} yr^{-1} in the top 5 cm of soil though the increase in TN was not significant.

Total Carbon

Paired comparisons of TC within Phase II sites indicated differing trends for each site (Table 3.2). ARM prairie and row crop pairs were not significantly different from each other though both contained significantly less TC than the corresponding control points. Within EIA samples, the row crop treatment contained significantly more TC than the PS treatment while the quantity of TC in control points was not significantly different from the row crop or PS samples. There was no significant difference in TC among RHO sample pairs.

Chronosequence sites did not follow a definite pattern through the whole timeline (Table 3.3). Interim 1 samples indicated a significant increase in TC from the row crop to 10 year prairie treatments. However, 25 and 37-year prairie sites did not contain a significantly different quantity of TC than either the row crop or 10 year prairie treatments. The difference in TC from the row crop and 37-year prairie indicated an average yearly increase in soil TC of 3.1 g C m^{-2} yr^{-1} though this change was not statistically significant.

Total Carbon to Total Nitrogen Ratio

Within Phase II sites, the TC:TN ratio varied by site and treatment (Table 3.2). At the ARM site, the control TC:TN ratio was significantly smaller than both the PS and row crop treatments. The TC:TN ratio of ARM PS and row crop were not significantly different In order from smallest to largest: PS, control, and row crop, EIA treatments were significantly different. There were no significant differences between RHO treatment TC:TN ratios. Chronosequence sites show a significant increase in TC:TN ratio with time in prairie (Table 3.3).

pH

Phase II sites were slightly acidic to neutral (Table 3.2). For both ARM and EIA sites, the pH of PS treatments were significantly greater than the pH of row crop treatments with no significant difference between the control samples with corresponding PS and row crop treatments. At the RHO site, the control samples had a significantly lower pH than the statistically similar PS and row crop treatments. In general, chronosequence sites exhibited a significant increase in pH with years since being in row crop (Table 3.3).

Bulk Density

Bulk density of the ARM Phase II site did not differ significantly by treatment pairs (Table 3.2). The PS treatment at the EIA site had significantly lower bulk density than both the EIA control and row crop treatments. Control samples at the RHO site had significantly greater bulk densities than the PS and row crop treatments. Bulk density decreased significantly with prairie age in the chronosequence sites (Table 3.3).

Infiltration

Field-saturated infiltration rates measured at the Phase II sites varied widely (Figure 3.8). Thus, statistics were ran as comparisons between paired treatments (row crop and PS) at each site prior to comparison among all Phase II sites by treatment. The Wilcoxon Signed Rank Test indicated the median difference between paired samples at ARM were not significant ($p=0.677$). ARM data was normally distributed with a wider range in infiltration rates among the PS compared to the row crop group (Table 3.4). RHO infiltration rates were similar in distribution (Figure 8) and not significantly different ($p=0.301$). The median difference between paired infiltration rates at EIA was significantly different ($p<0.05$). Infiltration rates at EIA among the PS samples vary significantly more widely than row crop infiltration. Overall, field-saturated infiltration from combined analysis across all three Phase II sites indicated the median difference among paired samples was not significant though infiltration in the PS treatments was greater than row crop treatments ($p=0.119$).

Aggregate Size Distribution

Aggregate size dependence on multiple factors including vegetation cover, time of year, and soil texture warranted comparison across similar soil types. Thus, comparisons were made between Phase II data (Figure 3.9), and data for chronosequence comparison (Figure 3.10) separately.

Among all Phase II treatments, the mass percentages of the >2 mm aggregate fractions were significantly larger than the other fraction mass percentages (Figure 3.9a). Within both ARM treatments, the 1-2 mm and 0.21-1 mm fraction percentages were significantly smaller than the >2 mm fraction percentage and significantly larger than the <0.21 mm fraction percentages. Compared between field treatment pairs with the same sieve size, the ARM row crop >2 mm aggregate fraction percentage was significantly larger than the corresponding PS fraction. Both ARM treatments for the 1-2 mm and <0.21 mm fraction percentages were not significantly different. The 0.21-1 mm PS fraction percentage was significantly larger than the row crop fraction.

Within EIA crop, the 0.21-1 mm and <0.21 mm fraction percentages were significantly smaller than the >2 mm fraction percentage and significantly larger than the 1-2 mm fraction percentage. In the EIA PS, there was no significant difference in the percentage of the fraction for the 1-2 mm, 0.21-1 mm, or <0.21 mm fraction percentages, though the fraction percentages were significantly smaller than the >2 mm fraction percentage. The >2 mm fraction in the PS treatment was significantly greater than the row crop fraction. The 1-2 mm fractions for both treatments were not significantly different. The remaining 0.21-1 mm and <0.21 mm fractions were significantly larger in the row crop fractions than the PS fractions.

The RHO PS aggregate fraction percentages for 1-2 mm, 0.21-1 mm, and <0.21 mm were statistically similar though they were significantly smaller than the >2 mm fraction percentages. Fraction percentage distributions among the RHO row crop treatment were the most widely varied of the treatments. The 1-2 mm fraction percentage was statistically the smallest for the RHO crop treatment among the fraction percentages. The 0.21-1 mm fraction percentage was significantly larger than the 1-2 mm fraction percentage and significantly smaller than the >2 mm and <0.21 mm fraction percentages. The <0.21 mm fraction percentage was significantly larger than the 1-2 mm and 0.21-1 mm and significantly smaller than the >2 mm fraction percentage. The >2 mm and 1-2 mm fraction percentages were significantly larger in the PS treatment compared to the row crop treatment. The 0.21-1 mm fraction percentage was not significantly different between treatments while the <0.21 mm fraction percentage in the row crop treatment was significantly larger than the corresponding PS fraction.

Chronosequence comparisons appeared to follow a more consistent trend than the paired sites (Figure 3.10a). Row crop and 10-year prairie treatments had >2 mm fractions significantly larger than 1-2 mm, 0.21-1 mm, and <0.21 mm fraction percentages. Additionally, the 25 and 37-year prairie treatments had significantly larger >2 mm fraction percentages compared to the smaller fraction classes. Both treatments had 1-2 mm fractions significantly larger than the <0.21 mm fraction percentages. The 0.21-1 mm fractions for 25 and 37-year prairies were not significantly different in quantity compared to the 1-2 mm and <0.21 mm fraction percentages.

Among >2mm fractions, the row crop treatment was significantly smaller than the 10, 25, and 37-year prairie treatments which were all statistically similar. The 1-2 mm fraction percentages for all treatments did not follow a definite increasing or decreasing trend with time. Both 0.21-1 mm and <0.21 mm fraction percentages were significantly larger in the row crop treatment compared to the similar prairie treatments.

Aggregate Fraction Carbon Content

Both ARM and RHO treatments as well as EIA row crop contained no significant difference between size classes and TC quantity (Figure 3.9b). Interestingly, the EIA PS sites showed significantly lower TC quantity in the 0.21-1 mm fraction than the 1-2 mm fraction. The >2mm and 1-2 mm fractions contained a similar quantity of TC. Between site treatment pairs, EIA TC in aggregate fractions were similar. The RHO PS treatment had significantly greater TC quantity in the 0.21-1 mm fraction compared to the row crop treatment while other fraction classes were similar. Within ARM treatments, TC quantity in >2 mm and 0.21-2 mm fractions were significantly greater in the PS treatment compared to row crop treatment. Chronosequence sites did not exhibit any significant differences within treatments (Figure 3.10b). Between treatments, TC quantity increased significantly with time since conversion to prairie for all aggregate size fractions.

Aggregate Fraction Nitrogen Content

Within treatments, both ARM and RHO sites did not contain significantly different quantities of TN by aggregate fraction (Figure 3.9c). The >2 mm EIA row crop aggregate fraction was similar in TN quantity to the other EIA row crop fractions. However, the 1-2 mm EIA row crop fraction contained a significantly greater quantity of TN than the corresponding

0.21-1 mm fraction. Within the EIA PS treatment, the >2 mm fraction and 1-2 mm fraction were similar and both fractions contained significantly more TN than the 0.21-1 mm fraction.

Between treatments, ARM aggregate fraction TN quantities were similar for the >2 mm and 1-2 mm fractions while the 0.21-1 mm fraction in the PS treatment contained significantly more TN than the corresponding row crop treatment. For all RHO aggregate fractions, TN content in the PS treatment was significantly greater than the row crop treatment. EIA aggregate fractions were mixed and the row crop treatment contained significantly more TN in the >2 mm and 0.21-1 mm aggregate fractions than the PS treatment. The 1-2 mm fractions contained similar TN quantities.

Chronosequence sites contained similar quantities of TN within treatments (Figure 3.10c). Similar to TC patterns, TN quantities appeared to increase significantly with time since conversion from row crop to PS.

Aggregate Fraction Carbon to Nitrogen Ratios

Phase II sites had similar TC:TN ratios within treatments for all sites (Figure 3.9d). Between ARM treatments compared by aggregate fraction size, the TC:TN ratio was not significantly different. For both EIA and RHO treatment comparisons, the 0.21-1 mm fraction in the PS treatment had significantly higher TC:TN ratios than the row crop comparison. For EIA and RHO, the >2 mm and 1-2 mm fractions were not different.

Chronosequence TC:TN ratios were not different within the 0, 10, 25, or 37-year treatments (Figure 3.10d). Over time, the TC:TN ratio did not significantly change.

Whole-Soil Particulate and Mineral Associated Organic Matter

Separation of particulate organic matter (POM) into two size fractions (coarse: 2-0.50 mm and fine: 0.50-0.053 mm) resulted in a coarse fraction with a smaller mass than required for analysis. The intention was to find TC and TN of the coarse POM by subtraction of fine POM and MAOM fractions from the whole-soil TC and TN. However, calculated differences had large errors with unrealistic TC and TN quantities. Thus, TC and TN contributions from the coarse POM fraction were omitted.

Phase II treatment pairs for fine POM-C indicated no significant differences at ARM and EIA sites (Table 3.5). The RHO PS treatment contained a greater amount of fine POM-C than the RHO row crop treatment. Fine POM-N values were similar within ARM treatments. The EIA row crop treatment had significantly greater fine POM-N content than the corresponding PS while RHO prairie had significantly greater fine POM-N content than the paired row crop site. Fine POM TC:TN ratios showed no significant difference between treatment at the Phase II sites.

Both MAOM-C and MAOM-N followed the same significant difference patterns. ARM treatments were not significantly different. EIA row crop contained greater concentrations of MAOM-C and MAOM-N than the paired PS treatment while RHO treatments were the opposite with greater concentrations of MAOM-C and MAOM-N in the PS treatment compared to the row crop treatment. Within Phase II sites, MAOM TC:TN ratios showed no significant differences within sites.

Trends within the chronosequence sites were mixed (Table 3.6). Frequently, significant changes were evident for the IN1 row crop and 10-year prairie treatments, but the addition of the 25 and 37-year sites did not always enhance the trend. For the full chronosequence, fine

POM-C increased significantly with time in prairie while fine POM-N increased initially then decreased back to lesser row crop concentrations. Similarly, MAOM-C concentrations appeared to remain steady while MAOM-N decreased significantly over the chronosequence timeline. Overall, fine POM and MAOM TC:TN ratios increased significantly.

Discussion

Nitrogen and Carbon Accumulation

Two years following conversion to prairie from row crop showed mixed results in nitrogen and carbon accumulation (Tables 3.2). However, in previous studies, there appeared to be a relationship with the higher TC content in chronosequences (Breuer, et al., 2006; Post and Kwon, 2000). A study of over 2000 soil pedons (primarily alfisols and mollisols) in Ohio suggested that soil taxon and drainage class accounted for the largest sources of variation followed by texture in the soil organic carbon pool among croplands and grasslands (Tan, et al., 2004). Thus, similarity of soil taxon may have been more important for carbon accumulation than soil clay content.

Soil taxon among the Phase II sites was accounted for by systematic sampling with PS, row crop, and control samples within soil types. Without direct analysis, drainage class was accounted for by the sampling strategy within each field by soil type. Thus, samples within fields were comparable, but comparison between sites without controlling for soil type variables may invalidate that comparison. The similarity among chronosequence soil types likely maintained that the sites were comparable.

An important factor in soil carbon accumulation, texture may have contributed to differences between the Phase II sites in TC and TN accumulation. Regardless of carbon input, clay soils have been shown to accumulate carbon quickly while sandy soils may hardly accumulate carbon after 100 years of inputs (Christensen, 1996). Studies indicated the largest soil TC pool was typically the mineral associated organic matter where carbon was adsorbed to clay and silt surfaces (Cambardella and Elliott, 1992; Hassink, 1997). Other TC pools within the soil were physically protected within microaggregates, biochemically held in complex compounds (i.e. lignin), and unprotected (Six, et al., 2002).

Given the higher sand and lower clay content at the EIA site, this may indicate a smaller capacity for carbon sequestration over time since carbon adheres to silt and clay particles. A simple regression of clay content versus TC at the EIA site did not produce a strong relationship between either the prairie or row crop treatment where R^2=0.02 and 0.08, respectively (data not shown). Further years of analysis would validate the capacity for carbon sequestration at sites with differing soil composition and precipitation.

Chronosequence TC accumulation was lower than may be expected compared to similar studies, though our measurements followed the expected trend of a high initial increase of soil TC in the early years followed by a lower rate of accumulation (Stewart, et al., 2007). The initial increase in TC for the first 10 years of 13 g m^{-2} is less than the 21 g m^{-2} measured in South Dakota (Post and Kwon, 2000). We may expect a higher rate of TC accumulation at the chronosequence sites given the higher quantity of precipitation in Iowa. However, soil TC capacity was not infinite. Thus, we may have reached an 'effective stabilization level' where TC inputs can no longer enhance soil content (Stewart, et al., 2007).

Conservatively, a mixture of cool and warm season grasses input 2900 kg C ha^{-1} yr^{-1} from dead roots and aboveground litter (Tufekcioglu, et al., 2003). In contrast, a corn and soybean rotation would add an average of 950 kg C ha^{-1} yr^{-1} from litter and dead roots (Tufekcioglu, et al., 2003). Differences in biomass inputs likely drive soil TC accumulations.

Nitrogen additions regulating carbon accumulation in prairie sites were likely to be primarily atmospheric deposition, microbial fixation, and redistribution (Knops and Tilman, 2000). Annual combined wet and dry nitrogen deposition measurements in Central Iowa were 0.771 g m^{-2} for 2003 (Anderson and Downing, 2006), and estimated oxidized N deposition from fossil fuel combustion was 0.62 g m^{-2} for the Mississippi River Basin (Howarth, et al., 2002b). Central Illinois deposition quantities for 2008-2011 ranging from 0.58-1.04 g m^{-2} indicate the potential for yearly deposition to be highly variable (Smith, et al., 2013). The chronosequence measurements from this study indicated a yearly increase of 0.45 g N m^{-2} in the first 10 years and an overall yearly increase of 0.01 g N m^{-2} for the 37-year period. Thus, without fixation or redistribution, deposition of nitrogen accounts for more TN than the soil accumulation. From the chronosequence perspective, this confirms our increase in soil nitrogen levels is realistic given environmental nitrogen contributions.

A vegetation assessment of the Phase II sites provided a qualitative review of potential for plant fixation. Vegetation surveys from the summer of 2016 indicated less than 50% of the vegetation at ARM and EIA were nonnative with ARM predominately forbs and EIA predominately grasses (Kordbacheh, unpublished data). Greater than 75% of the species at RHO were nonnative grasses (Kordbacheh, unpublished data). The prevalence of forbs and potential for nitrogen fixation from legumes may have contributed to soil nitrogen content at the ARM prairie site while the predominately grass populations would not have added to the nitrogen pools via fixation at the EIA and RHO sites. However, a regression developed in the

Knops and Tilman (2000) study indicated legume presence in perennial vegetation did not significantly impact soil carbon, nitrogen, or TC:TN.

At some Phase II sites, nitrogen content decreased following conversion to prairie though the change was not always significant. A similar yearly trend was observed in Central Minnesota where the change in soil nitrogen content following row crop to prairie conversion ranged from -0.15 to 1.93 g m^{-2} with an average yearly gain of 1.23 g N m^{-2} yr^{-1}. (Knops and Tilman, 2000). Thus, an early depletion of the nitrogen pool is not unusual.

Based on previous research, we expect the soil carbon and nitrogen cycles to be tightly coupled (Breuer, et al., 2006; Jensen, 1997) with carbon accumulation controlled by nitrogen accumulation. Additionally, immobilization of TN fueled by surplus TC appears to protect against TN losses via leaching (Schipper, et al., 2004). Consequently, N mineralization depends on the TC:TN ratio which drives TC and TN accumulation within the soil profile (Springob and Kirchmann, 2003).

For the chronosequence series, we did see an increase (though not significant) over time in the TC:TN ratio (Table 3.3) which had been reported in other chronosequence studies, (Breuer, et al., 2006; Knops and Tilman, 2000). It was interesting to note that while the overall TC:TN ratio did not change, the POM and MAOM TC:TN ratios increased with time in prairie vegetation indicating as TC:TN ratios increase, there is more C per unit N so the N is more tightly held and limiting to plant growth.

Infiltration

There was no clear relationship between soil texture and infiltration rates among Phase II sites. Antecedent moisture content was not quantified prior to infiltration measurements, but should not affect steady state infiltration rates as long as the soil was not saturated. At the time of infiltration measurements, PS had been in place for 2 (ARM and EIA) or 3 (RHO) years. Subsequent years with PS treatment in place will likely increase infiltration as larger soil aggregates form (Bharati, et al., 2002; Le Bissonnais, 1996). An additional impact on infiltration in row crop treatments, compaction from wheel tracks, would reduce infiltration and increase bulk density (Alizadehtazi, et al., 2016; Håkansson, et al., 1988). For this study, visibly compacted row spaces were carefully avoided. A decrease in bulk density within the PS sites may indicate an expected result of the conversion to prairie, but if the track rows were not avoided, this difference in bulk density and thus infiltration by treatment would be exacerbated.

An infiltration study in Northern Missouri supported the decision to not measure infiltration at long-term prairie sites as runoff may not be achieved with the infiltrometer method (Anderson, et al., 2009). The recommended rainfall rate for infiltrometers range from 20-30 cm hr^{-1} which may not be high enough to measure runoff at long-term perennial vegetation sites (Bharati, et al., 2002).

Aggregate Size Distribution

Timing for aggregate size sampling was imperative as soil aggregate stability and size distribution within fields vary by season, recent temperatures, and moisture (Lehrsch and Jolley, 1992; Mulla, et al., 1992). Aggregate stability has been shown to decrease significantly from October to March (Harris, et al., 1966). Thus, fall post-harvest sampling was time sensitive. Phase II sites were sampled within less than a month from each other in late October and mid-November. IN1, IN4, and KRU sampling was completed in early December and may contribute to the apparent though insignificant decline in the >2 mm fraction mass with increasing years in prairie.

A study reviewing a series of pasture experiments reported wet sieving did not break apart a significant number of aggregates with 90% of the soil in >0.250 mm fractions (Gijsman and Thomas, 1995). Additionally, air-drying the aggregates has been shown to increase stability of aggregates fractions (Reid and Goss, 1981) and wetting to field capacity plus 5% results in more stable aggregates (Márquez, et al., 2004). Thus, this study may have masked some of the variation that would be apparent if samples had been wet sieved at field moisture, or slaked when rewetting (Cambardella and Elliott, 1993).

Significant differences in aggregate size distribution and nutrient content between treatments at each Phase II field site may indicate that 2 years post-conversion is an adequate time frame to quantify transitional differences at these fields. Given the different locations for chronosequence sampling, it is important to note that without initial soil property quantification prior to conversion, higher TC and TN measurements may be a result of higher soil nutrient content at the time of conversion (Knops and Tilman, 2000).

An additional variable to review among the Phase II sites was tillage. Tillage may partially explain the larger <0.21 mm fractions at EIA and RHO sites though the soil hadn't experienced tillage in almost a year. In contrast, the ARM site had larger 1-2 mm and 0.21-1 mm aggregate fractions than both EIA and RHO while experiencing no annual tillage.

Despite the mass emphasis of the aggregate size distribution on the larger fractions, there was not a difference within treatments for the chronosequence sites on the concentration of TC and TN. This may indicate organic matter content was related to aggregate size distribution (Cambardella and Elliott, 1993).

Whole-Soil Particulate and Mineral Associated Organic Matter

As POM becomes further decomposed, the TC:TN ratio decreases (Parker, 1962). It has been shown that mineral-associated organic matter (MAOM) saturated by adsorbing to clay and silt particles while changes in soil carbon were associated with larger soil particles and the addition of particulate organic matter (Hassink, 1997). The underlying mechanism for MAOM saturation is expected to be physical protection of organic matter from silt and clay particles (Theng, 1979).

We may not expect similar TC:TN ratios in MAOM by field for the Phase II treatments since soil particle size affected capacity for adsorption (Hassink, 1997; Zhang, et al., 1988). The Zhang et al. (1988) study in Central Iowa indicated MAOM TC:TN in agricultural fields were near 10 like our Phase II results (Table 3.5). Given the similarity in soil types, we would expect to see similar TC:TN for MAOM between chronosequence sites. Our results indicated the potential for an increased TC:TN ratio in the MAOM fraction 37 years post conversion to prairie as the fraction became more saturated (Stewart, et al., 2007).

Particulate organic matter was composed primarily of partially decomposed root fragments (Cambardella and Elliott, 1993). POM served as a labile carbon pool (Hassink, 1997), and POM-C was biologically available for microorganisms and important for nutrient cycling (Marquez, et al., 1998). Thus, sites with greater POM concentrations in the soil may expect better soil nitrogen retention and cycling.

Based on the similar whole-soil-C and MAOM-C pools for the chronosequence fields (Table 3.6), it was interesting to note differences in POM-C potentially indicating an initial bump followed by a general decrease or leveling off. The difference within IN1 by treatment indicated an increase in POM following conversion to prairie within the same field. Given the opportunity for future sampling, it remains to be determined if that fraction has reached equilibrium. Without supplementary data from local chronosequences, an assumption of equilibrium is a risky conclusion. Strategic resampling in 10 years would further develop the Phase II and chronosequence data sets to assess pool changes (if any) and the potential for carbon accumulation.

Conclusions

Quantification of numerous soil parameters highlighted the variability among the current condition of Iowa soils. The opportunity to present expected changes in soil properties following land use modification would be useful for estimating the shift in soil nutrient content, infiltration, and capacity for nutrient cycling. Results from sites 2 years post conversion to row crop served as caution for extrapolating results from each site given the regional variability. Thus, future sampling will be required at or within similar soil types at each of the Phase II sites.

The chronosequence provided an overview of the expected transformation timeline for row crop reverted to prairie. Within 37 years, whole soil carbon and nitrogen did not accumulate significantly. However, aggregate fraction TC and TN accumulated significantly as did POM-C. This may indicate that soil TC accumulated in stages with prairie litter and POM inputs. Aggregates formed around POM and physically protected it from decomposition. Average carbon accumulation of 3.1 g C m^{-2} yr^{-1} indicated reverted prairies are carbon sinks. TN did not accumulate except at the 10-year prairie site which was adjacent to row crop and may receive some supplemental nitrogen inputs from shallow groundwater.

Soil aggregate size distribution and particulate organic matter were quantified within the Phase II and chronosequence sites separately given the propensity for the effect of soil types to affect soil aggregation and the quantified differences in soil properties. In general, Phase II sites did not exhibit clear patterns in aggregate nutrient content, although the >2 mm fraction was consistently the largest across all treatments and fields. Trends within POM-C and POM-N at Phase II sites did not show a general increase or decrease among all sites between treatments.

In contrast to the general unclear trend of the Phase II sites, chronosequence results for aggregate size distribution indicated a significant increase in carbon and nitrogen content among aggregate fractions with no significant change in the TC:TN ratio. POM-C and POM-N trends between the chronosequence sites appear to increase initially among the IN1 0 and 10 year treatments then level off with time. This may indicate an unaccounted for fundamental difference between field locations (Breuer, et al., 2006; Lal, 2002). Neither MAOM pool increased steadily in the chronosequence sites. However, both POM and MAOM TC:TN ratios increased significantly suggesting the MAOM pool could be TN saturated and lacking in TC.

Implementation of sampling at the Phase II sites would be useful to develop regional chronosequences and clarify soil property changes following the conversion of row crop to prairie vegetation. Regional factors like soil texture and precipitation may change the timeline for nutrient accumulation and thus local comparisons are important (Lal, 2002). While the chronosequence presented in this study details soil property changes following conversion to prairie, the lack of initial measurements of soil properties prior to conversion may mask soil property changes that were significant. Future review of the Phase II and chronosequence sites would serve to further inform soil property trends.

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Table 3.1. Number of sample points per treatment and soil characteristics for each field

Field	Prairie Age	Tillage	2016 Crop Rotation	Samples per Treatment			Number of Soil Types	Particle Size		
				Prairie	Row Crop	Control		Sand (%)	Silt (%)	Clay (%)
Armstrong	2	No	Soybean	12	12	6	4	14	56	30
East Iowa Airport	2	Yes	Soybean	18	18	15	6	26	46	27
Rhodes	2	Yes	Corn	15	15	12	5	17	52	31
Interim 1	10	No	Corn	9	9	0	3	14	46	40
Interim 4	25	NA	NA	9	0	0	3	17	47	36
Krumm	37	NA	NA	9	0	0	3	20	49	31

Note: Prairie Age describes the number of years since conversion from row crop to prairie

Table 3.2. Phase II field whole-soil properties and significance levels ($p<0.10$)

Field	Treatment	Prairie Age	pH	Bulk Density ($g\ cm^{-3}$)	TC ($g\ m^{-2}$)	TN ($g\ m^{-2}$)	TCTN
Armstrong	Control	0	6.8 AB	1.02 A	1538 A	174 A	9 B
Armstrong	Prairie	2	7.2 A	1.04 A	1191 B	111 B	11 A
Armstrong	Row Crop	0	6.9 B	1.08 A	1162 B	115 B	10 A
East Iowa Airport	Control	0	6.5 AB	1.11 A	1350 AB	120 A	11 B
East Iowa Airport	Prairie	2	6.8 A	0.94 B	1144 B	111 A	10 C
East Iowa Airport	Row Crop	0	6.4 B	1.11 A	1413 A	116 A	12 A
Rhodes	Control	0	6.8 B	1.15 A	872 A	87 A	10 A
Rhodes	Prairie	2	7.0 A	1.03 B	851 A	85 A	10 A
Rhodes	Row Crop	0	7.0 A	0.99 B	924 A	91 A	10 A

Note: Significant differences between treatments are marked with different letters. Paired comparisons between treatments were made within the same field.

Table 3.3. Chronosequence field soil properties and significance levels ($p<0.10$)

Field	Treatment	Prairie Age	pH	Bulk Density (g cm^{-3})	TC (g m^{-2})	TN (g m^{-2})	TCTN
Interim 1	Row Crop	0	6.1 C	1.15 A	1567 B	139 A	11.3 B
Interim 1	Prairie	10	6.6 B	1.04 B	1691 A	143 A	11.9 AB
Interim 4	Prairie	25	6.7 A	1.05 B	1676 AB	138 A	12.2 A
Krumm	Prairie	37	6.5 B	0.85 C	1682 AB	139 A	12.2 A

Note: Significant differences between soil properties based on prairie age are marked with different letters.

Table 3.4. Phase II field infiltration summary for 2-year paired prairie sites

Field	Treatment	n	Mean Field Saturated Infiltration (cm min^{-1})	Standard Deviation	COV	Median (95% confidence interval)
Armstrong	Row Crop	12	0.13	0.07	53.7	0.12 ($0.09 \leq x \leq 0.17$)
Armstrong	Prairie	12	0.16	0.12	73.8	0.16 ($0.08 \leq x \leq 0.24$)
East Iowa Airport	Row Crop	18	0.02	0.02	129.4	0.01 ($0.01 \leq x \leq 0.03$)
East Iowa Airport	Prairie	18	0.04	0.04	116.9	0.02 ($0.02 \leq x \leq 0.06$)
Rhodes	Row Crop	9	0.01	0.01	64.8	0.01 ($0.00 \leq x \leq 0.02$)
Rhodes	Prairie	9	0.02	0.01	70.8	0.02 ($0.01 \leq x \leq 0.03$)
Overall Phase II Sites	Row Crop	39	0.05	0.07	132.1	0.03 ($0.03 \leq x \leq 0.07$)
Overall Phase II Sites	Prairie	39	0.07	0.09	129.3	0.04 ($0.04 \leq x \leq 0.10$)

Table 3.5. Carbon and nitrogen pools for Phase II fields in the top 5 cm of soil

Field	Treatment	Whole Soil	Fine POM	MAOM	Whole Soil	Fine POM	MAOM	Fine POM	MAOM
		-------------g C m^{-2}-------------			-------------g N m^{-2}-------------			TC:TN	
Armstrong	Row Crop	1162 A	231 A	820 A	115 A	14 A	90 A	16 A	9 A
Armstrong	Prairie	1191 A	233 A	840 A	111 A	14 A	89 A	16 A	9 A
East Iowa Airport	Row Crop	1413 A	139 A	1027 A	116 A	11 A	91 A	13 A	11 A
East Iowa Airport	Prairie	1144 A	116 A	809 B	111 A	10 B	72 B	12 A	11 A
Rhodes	Row Crop	924 A	226 B	627 B	91 A	14 B	72 B	16 A	9 A
Rhodes	Prairie	851 B	329 A	748 A	85 A	20 A	84 A	17 A	9 A

Note: Significant differences between paired field treatments are marked with different letters

Table 3.6. Carbon and nitrogen pools for chronosequence fields in the top 5 cm of soil

Field	Whole Soil	Fine POM	MAOM	Whole Soil	Fine POM	MAOM	Fine POM	MAOM
	-------------g C m^{-2}-------------			-------------g N m^{-2}-------------			TC:TN	
IN1 Row Crop	1567 B	220 B	1148 AB	139 A	13 B	109 A	17 C	11 C
IN1 10-year Prairie	1691 A	326 A	1210 A	143 A	17 A	109 A	20 B	11 BC
IN4 25-year Prairie	1676 AB	301 A	1184 AB	138 A	12 B	105 A	25 A	11 AB
KRU 37-year Prairie	1682 AB	304 A	1124 B	138 A	13 B	95 B	23 A	12 A

Note: Significant differences between treatments are marked with different letters

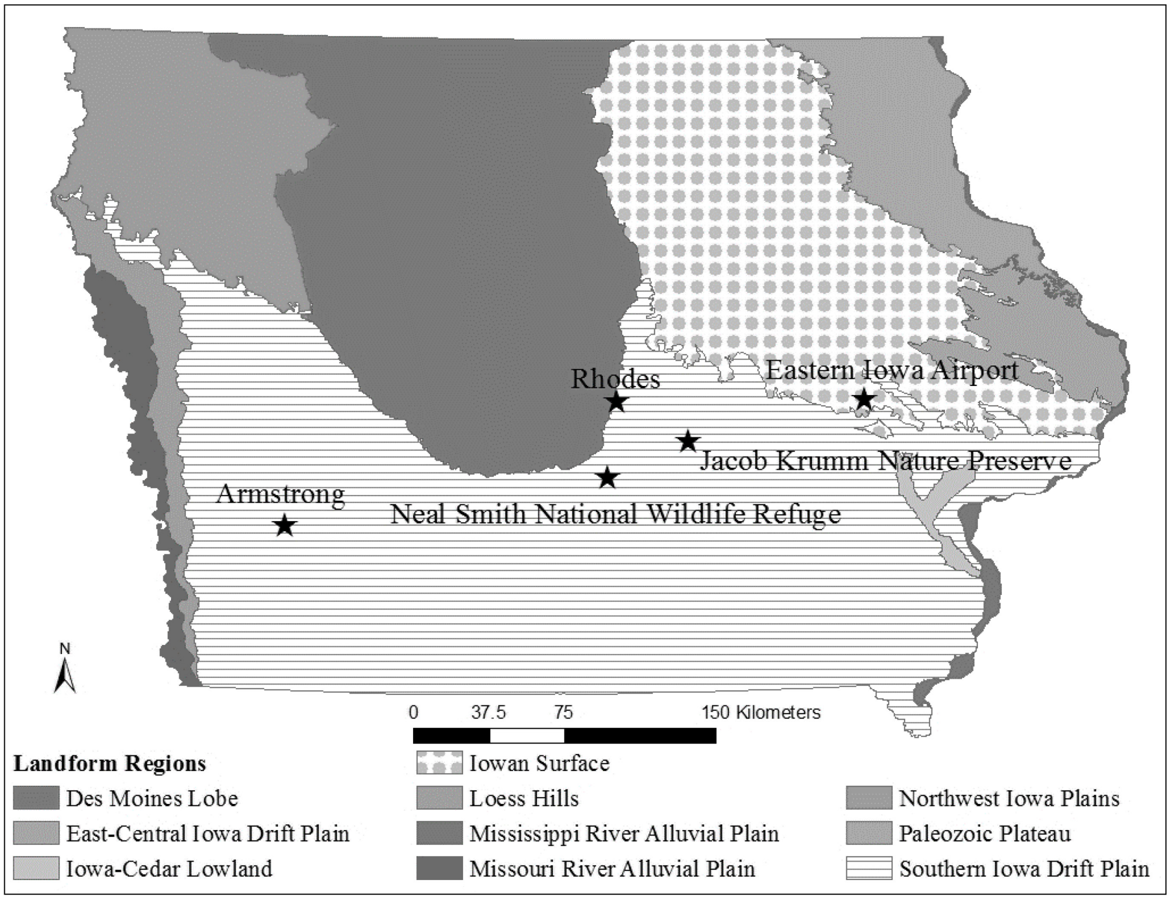

Figure 3.1. Iowa landform regions and field sites

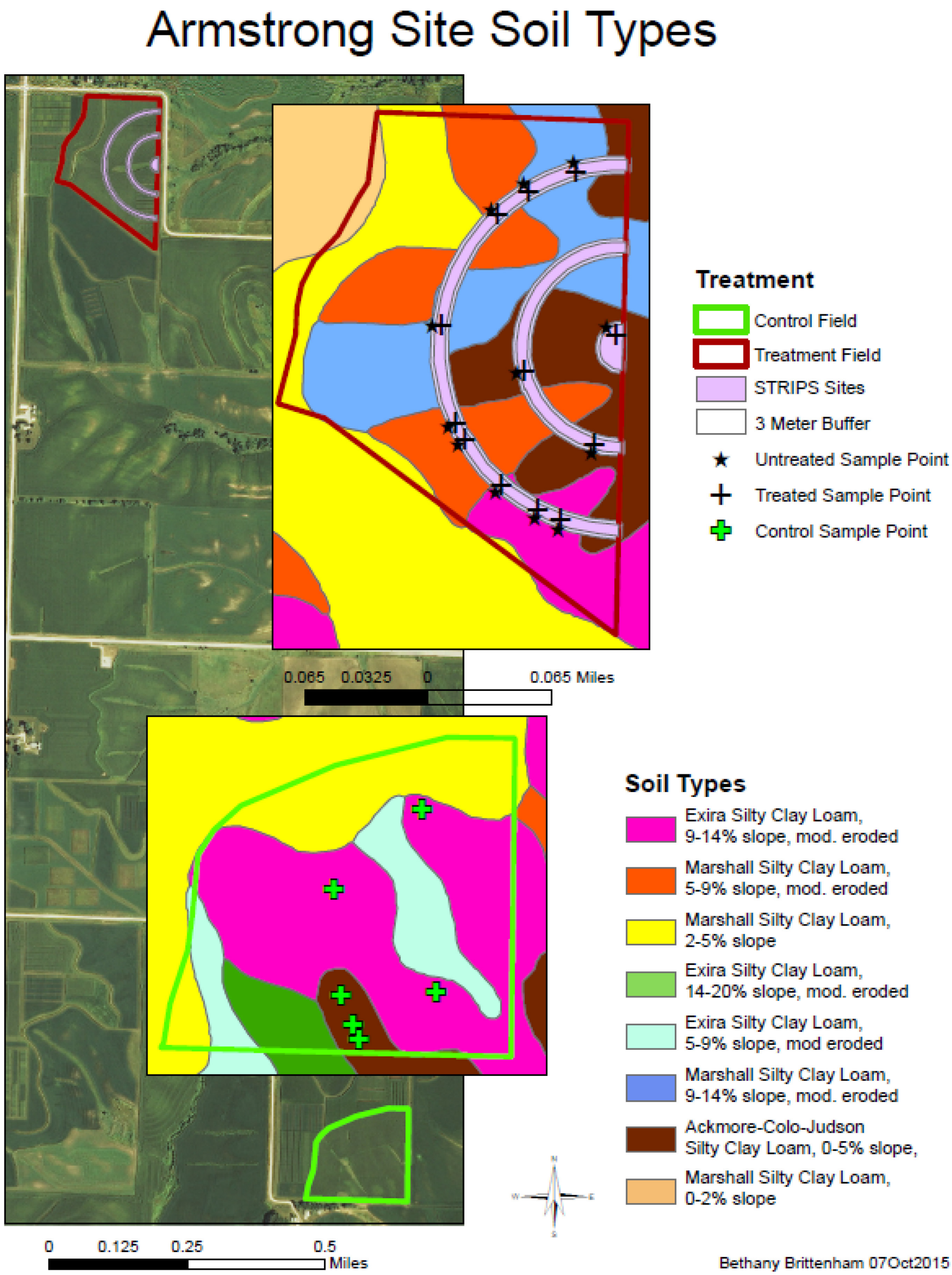

Figure 3.2. Armstrong field site with soil types and sample points

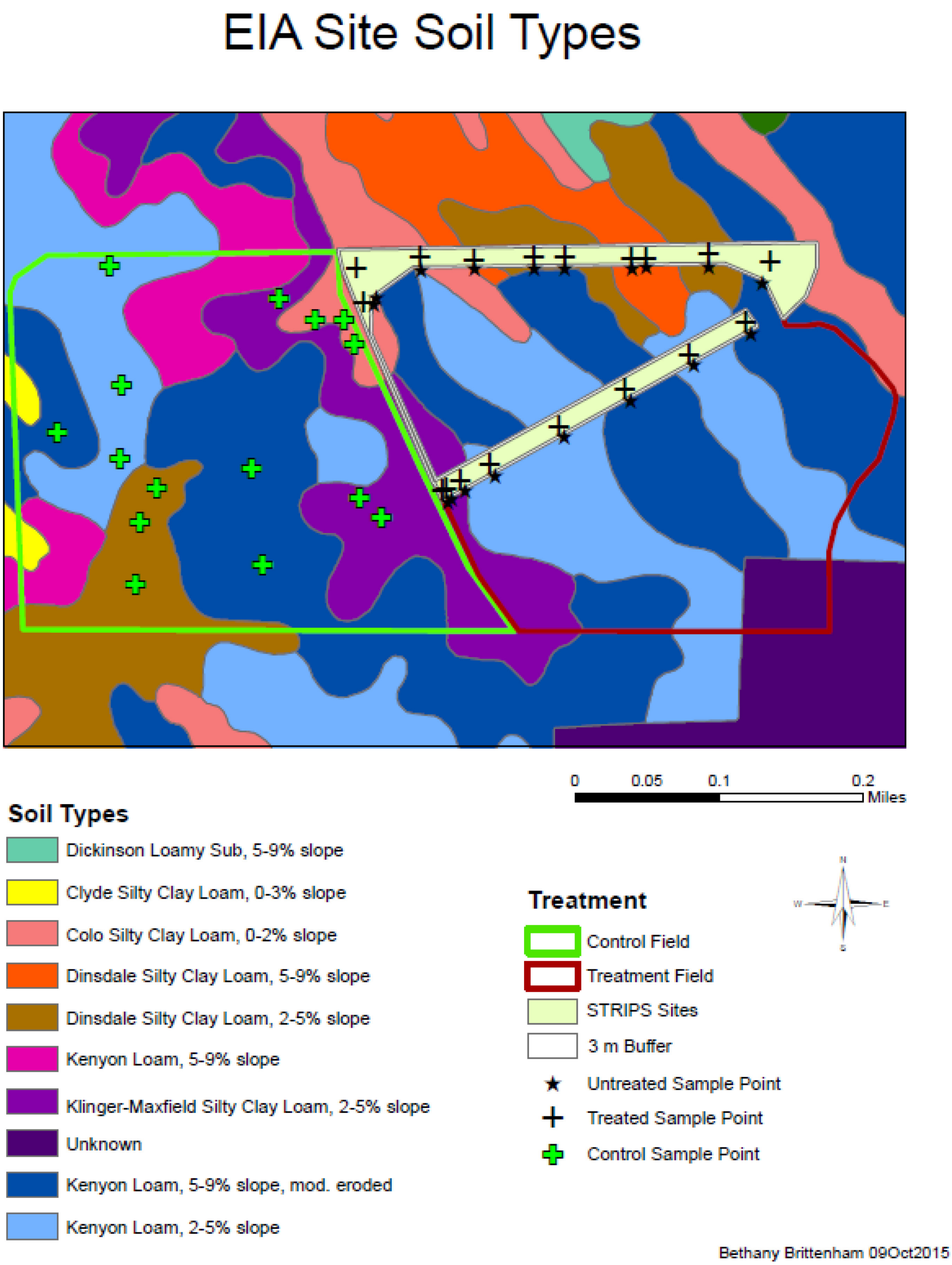

Figure 3.3. EIA field site with soil types and samples points

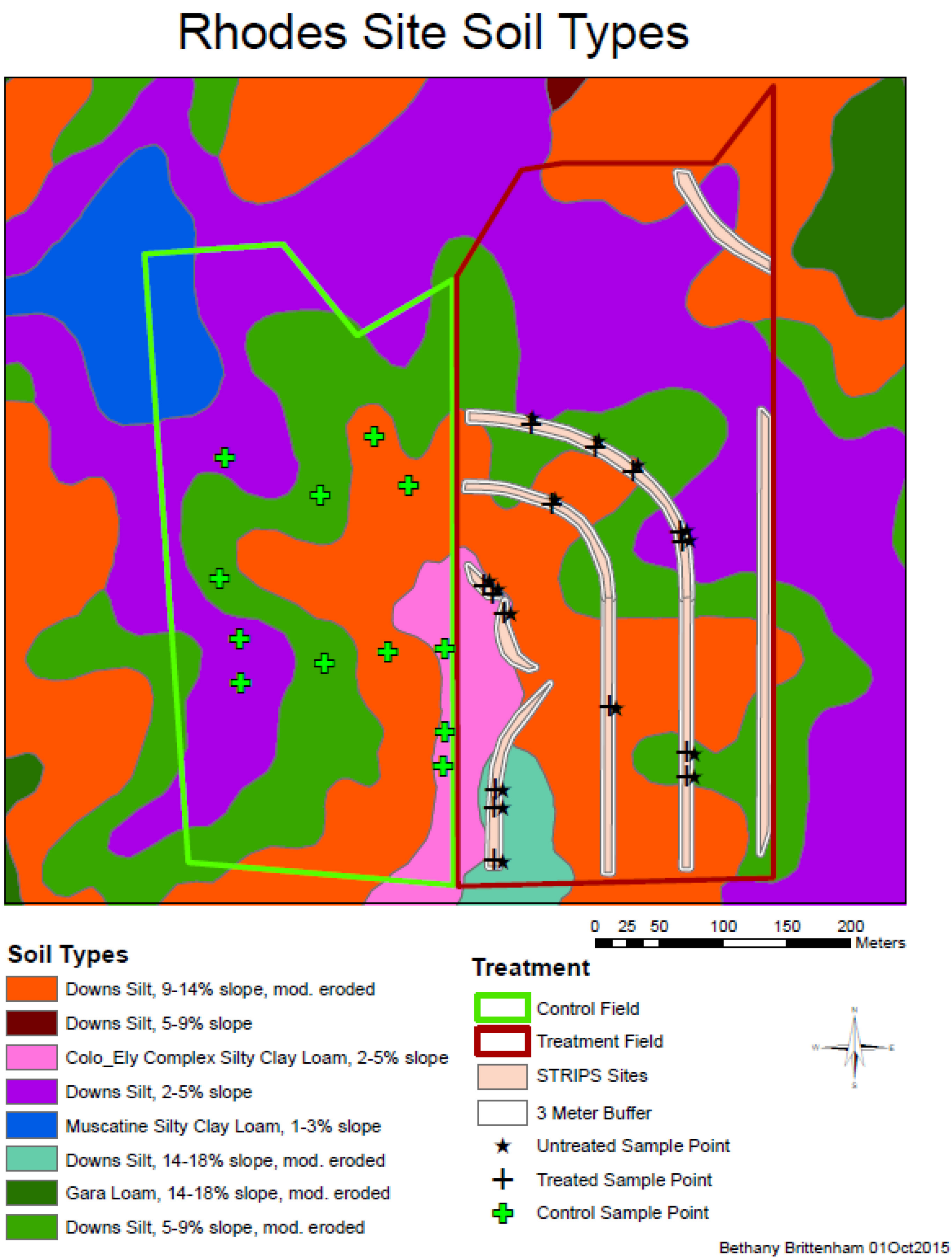

Figure 3.4. Rhodes field site with soil types and sample points

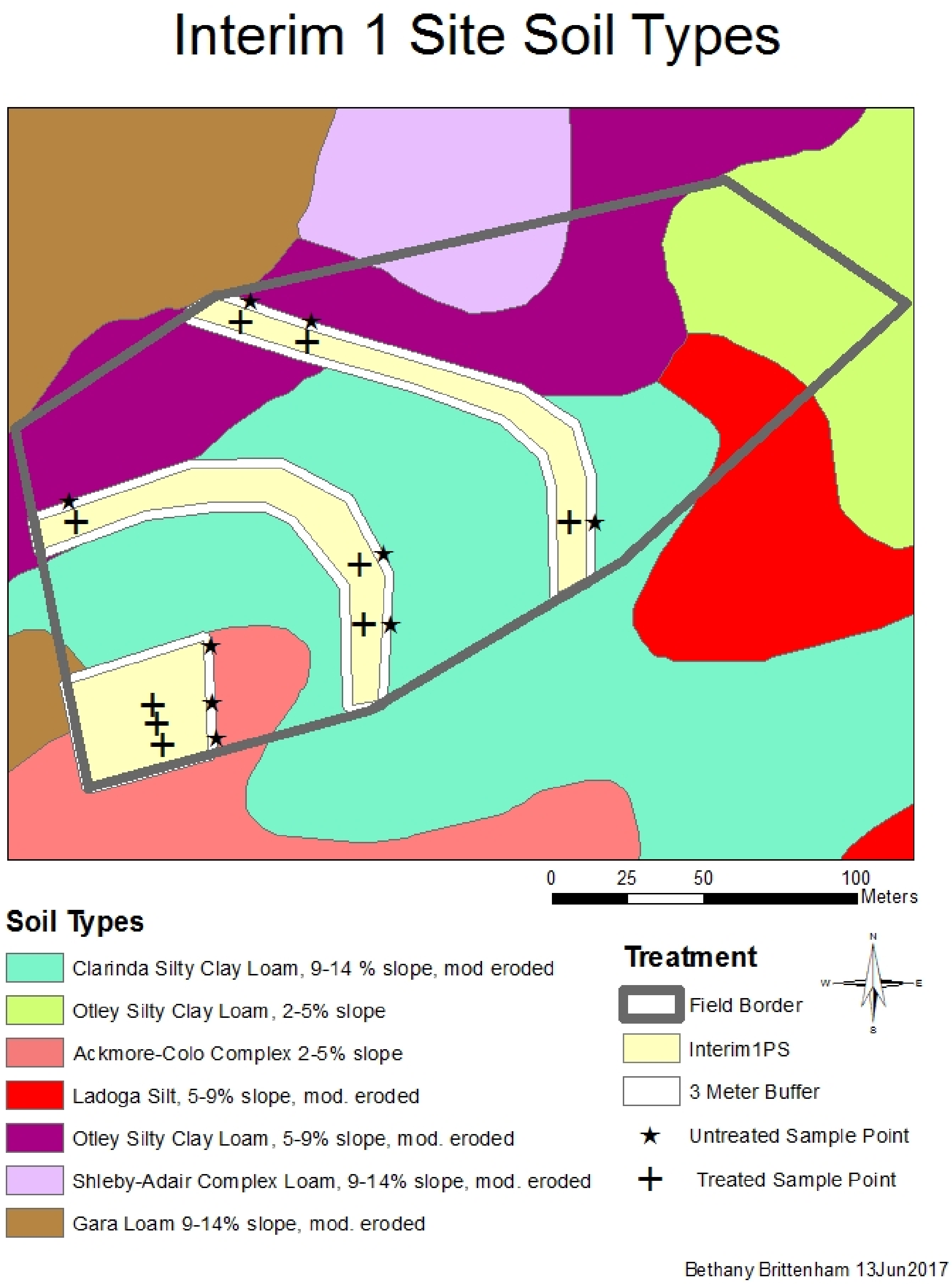

Figure 3.5. Interim 1 field site with soil types and sample points

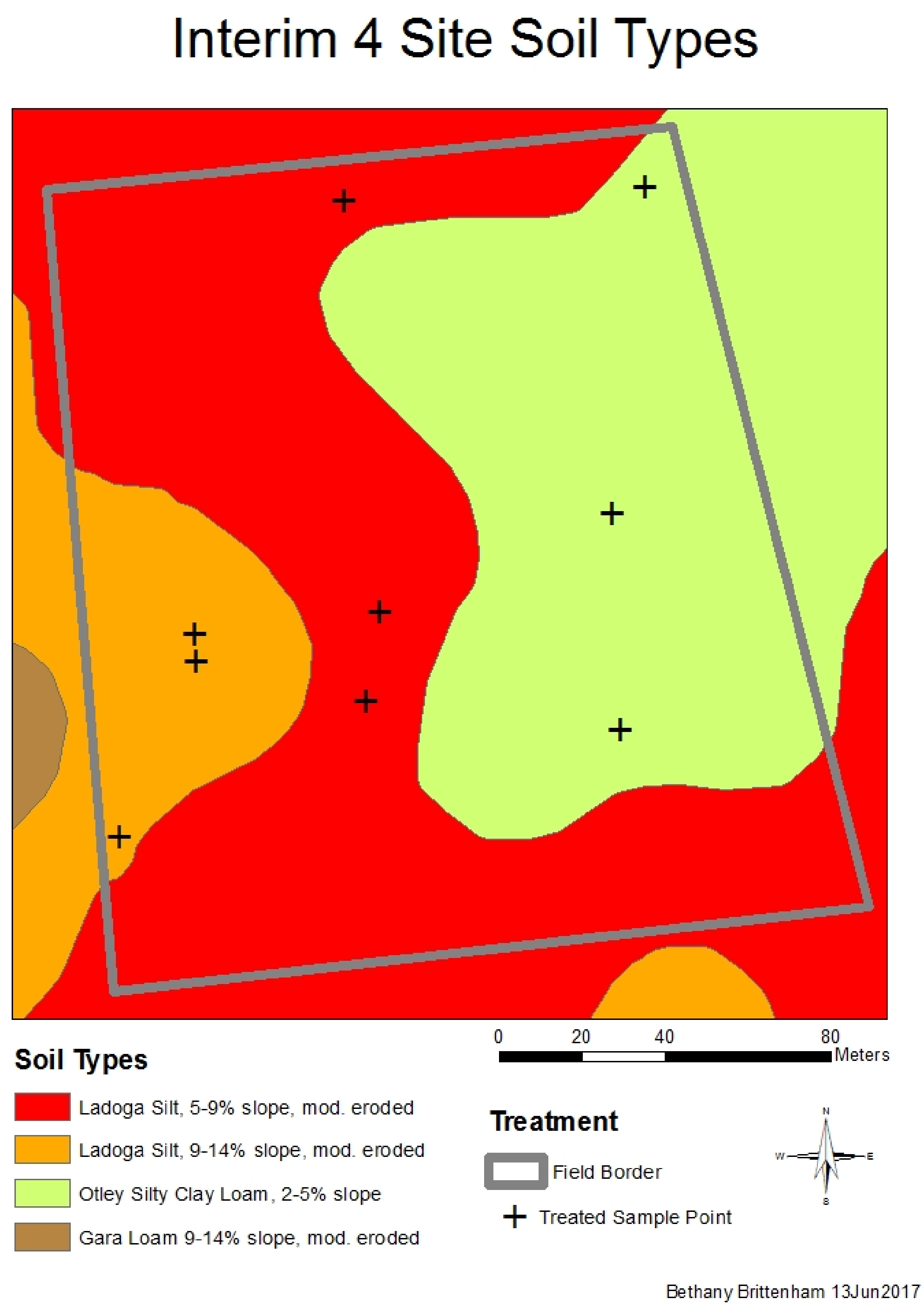

Figure 3.6. Interim 4 field site with soil types and sample points

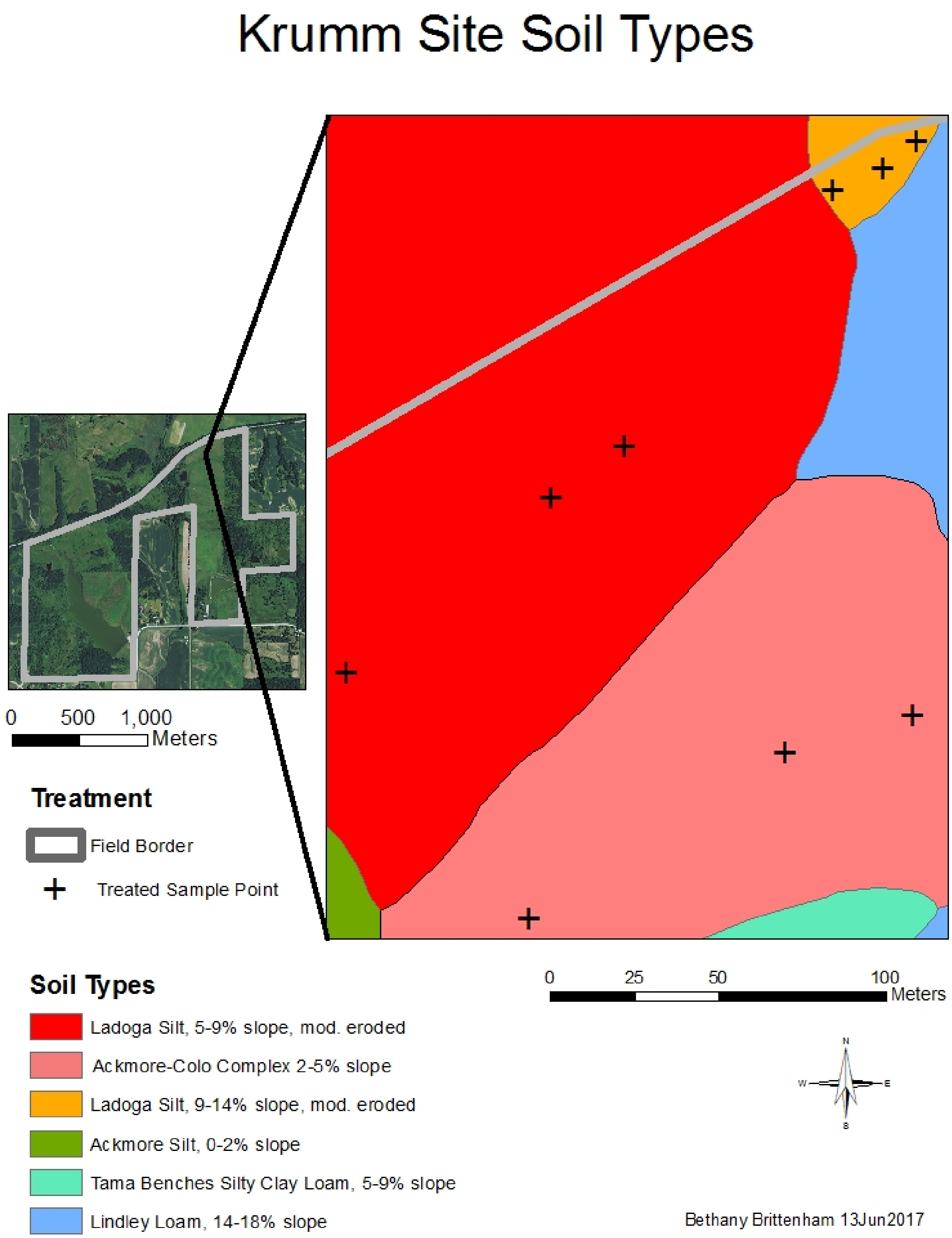

Figure 3.7. Krumm field site with soil types and sample points

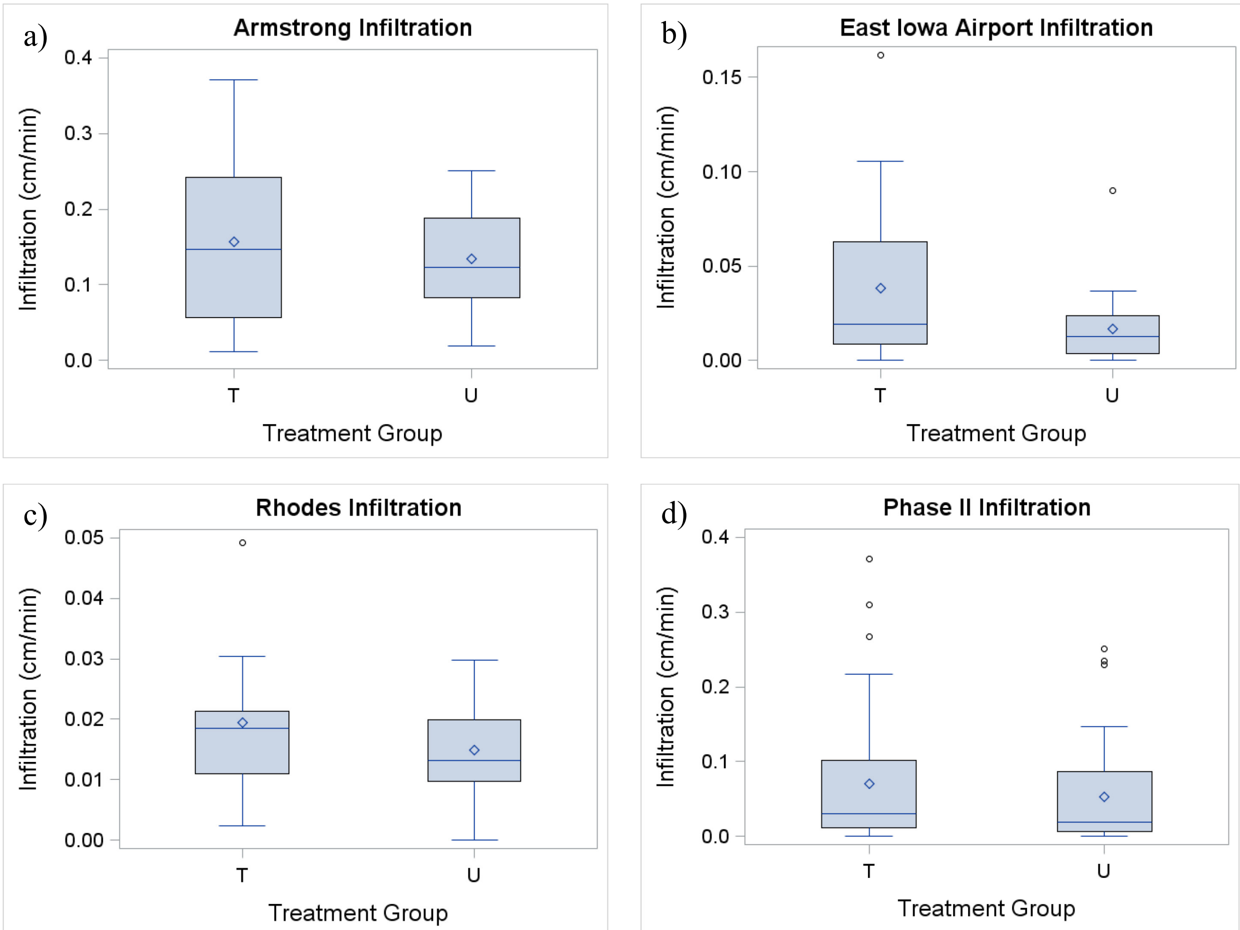

Figure 3.8. Field infiltration by treatment for Phase II sites.

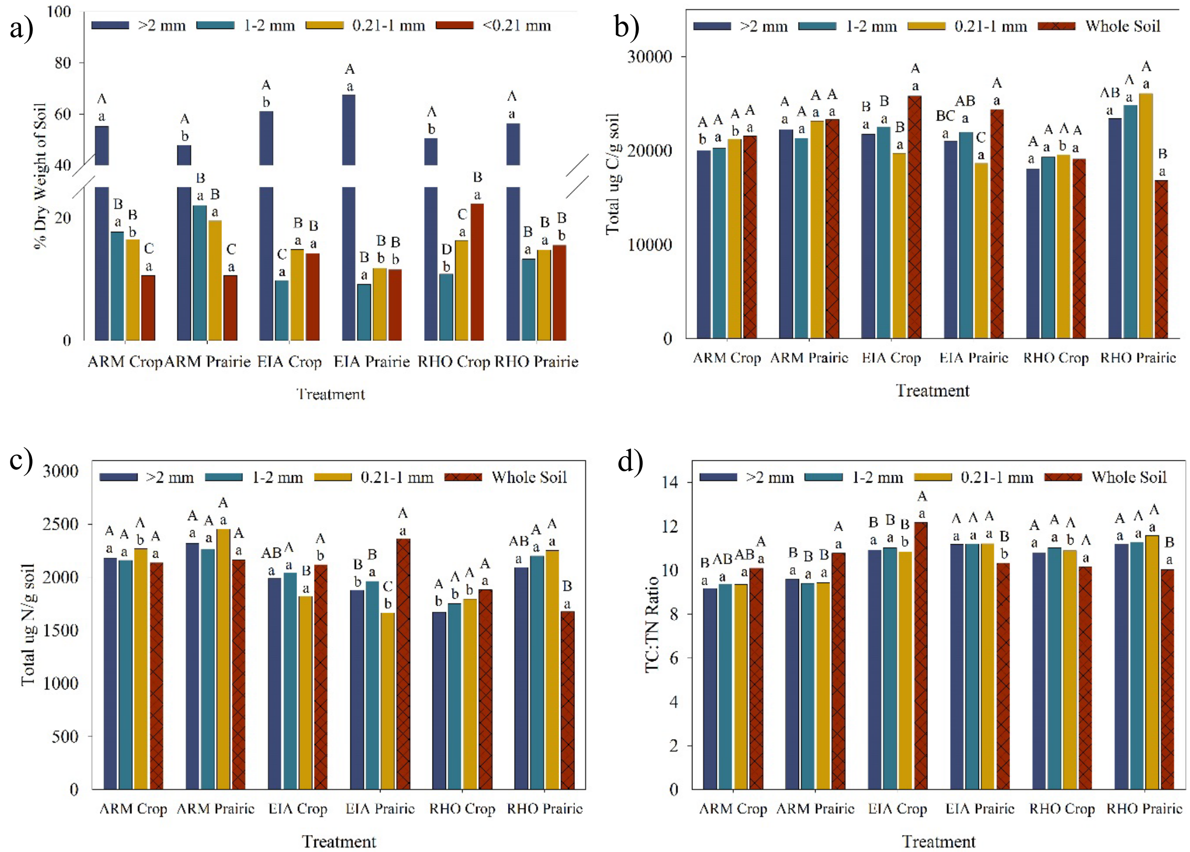

Figure 3.9. Comparisons among Phase II sites for a) aggregate size distribution, b) aggregate carbon content, c) aggregate nitrogen content, and d) TC:TN ratio by fraction. Different uppercase letters indicate significant ($p<0.10$) differences within treatments and between aggregate size fractions. Different lowercase letters indicate significant ($p<0.10$) differences within aggregate size fractions between treatments.

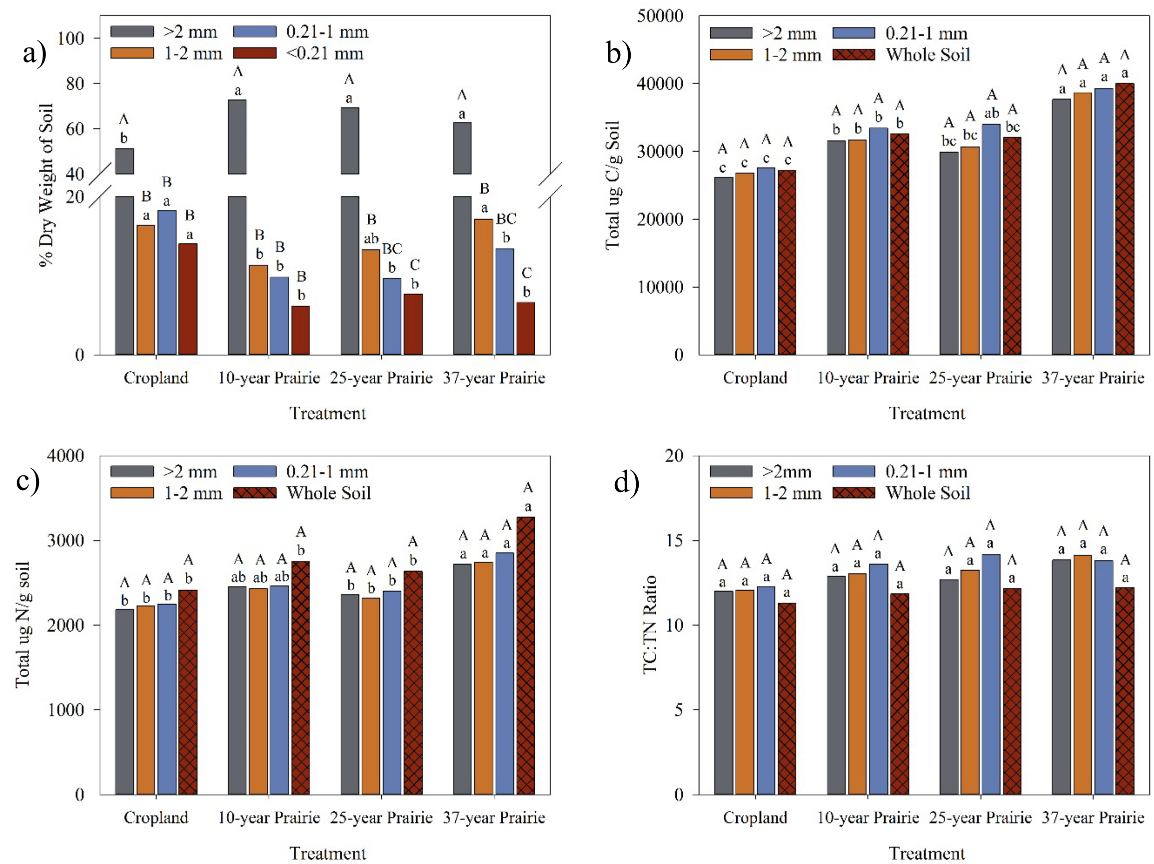

Figure 3.10. Comparisons among chronosequence sites for a) aggregate size distribution, b) aggregate carbon content, c) aggregate nitrogen content, and d) TC:TN ratio by fraction. Different uppercase letters indicate significant ($p<0.10$) differences within treatments and between aggregate size fractions. Different lowercase letters indicate significant ($p<0.10$) differences within aggregate size fractions between treatments.

CHAPTER 4. CONCLUSIONS

General Discussion

Concentrations of nitrate-nitrogen in shallow groundwater within the top 2 meters of the soil surface decreased following implementation of prairie strips (PS) in row crop fields regardless of PS placement. However, PS placement at the footslope only instead of including contour strips of vegetation increased dissolved phosphorous concentrations in groundwater likely due to shallow water tables and denitrifying conditions that make phosphorous more soluble. Both the 10% contour strip and 20% contour strip PS layouts appear to be the most effective at reducing nutrient export via shallow groundwater.

Quantification of soil property changes with a 37-year chronosequence for row crop to prairie conversion offered insight into how soil accumulated carbon and nitrogen. Overall increase in POM-C and aggregate TC may indicate prairie biomass inputs added to those pools prior to significantly enhancing whole soil TC. Macroaggregates likely developed around POM and physically protected the biomass from degradation. TN did not accumulate significantly in any pools except in POM at the 10-year prairie site. Samples from the 10-year prairie may have received external TN inputs from the adjacent row crop. Overall, the trends depicted in chronosequence results may foreshadow similar changes to be expected from the 2-year sites with varying soil types. However, without direct quantification, assumptions should not be made on the change in soil properties.

Recommendations for Future Research

Both studies within this thesis highlighted the need for further research on the topics of nutrient content in shallow groundwater with prairie strips (PS) and the modification of soil properties following conversion from row crop to prairie vegetation:

1. Quantify treatment effect of PS in regions with deeper water tables where nitrate-nitrogen would likely leach deeper than the 2 meter treatment zone.
2. Current yearly management of established PS is mowing and removal of vegetation, and the effect of alternative PS management methods like controlled burning or grazing was not taken into consideration in terms of nutrient concentrations in shallow groundwater.
3. Future soil sampling at the Phase II sites would enhance the dataset for expected regional modifications in soil properties following conversion to prairie since the current 2-year post conversion soil measurements did not indicate a clear trend.
4. Additional future sampling at the chronosequence sites would boost current soil property data and clarify uncertainties in particulate and mineral-associated organic matter trends.